7 Health Effects of Particulate Matter

The Children's Environmental Health Protection Act (Senate Bill 25, Senator Martha Escutia, Stats. 1999, Ch. 731) required the ARB, in consultation with OEHHA, to "review all existing health-based ambient air quality standards (AAQS) to determine whether, based on public health, scientific literature, and exposure pattern data, these standards adequately protect the health of the public, including infants and children, with an adequate margin of safety." Of those AAQS identified as providing insufficient public health protection, SB 25 requires the ARB to revise the highest priority standard by December 31, 2002. Last year OEHHA staff, assisted by six academic air pollution researchers, undertook a critical review of the health impacts of exposure to the regulated pollutants, and categorized the latter into two tiers, with the first representing greater potential risks to public health at the concentrations of the current AAQS. Of the first-tier standards, OEHHA identified the AAQS for particulate matter as the highest priority pollutant, and recommended to the ARB that this standard be the first to consider for a more thorough evaluation and possible revision. This decision was based on the evidence in the literature of health effects, including mortality and morbidity in infants, children and other potentially sensitive subgroups, associated with particulate matter at or below the current state standards. The ARB accepted the recommendation by OEHHA staff at the Board Meeting held December 2000.

This chapter contains a targeted, critical review by OEHHA staff of the research relevant to setting the standard(s) for the particulate matter AAQS for California. Beginning with dosimetry of particles (Section 7.1), the review focuses primarily on epidemiological studies of mortality associated with both acute and chronic exposure to PM (Sections 7.2, 7.3 and 7.4), as well as morbidity outcomes (Sections 7.5 and 7.6). This review of the most pertinent literature is followed by discussions of susceptible subpopulations (Section 7.7), plausible biological and toxicological mechanisms underlying the epidemiological observations (Section 7.8), and causal inference regarding the associations between ambient PM concentrations and increased morbidity and mortality (Section 7.9). An assessment of the potential benefits associated with reducing exposures to ambient PM is presented in Section 7.10, followed by the OEHHA staff recommendations for revision of California's AAQS for PM (Section 7.11).

In brief, OEHHA staff recommends that the current PM10 annual average standard be revised. There are compelling reasons to be concerned about significant adverse health effects associated with ongoing exposures occurring at or below concentrations prescribed by the existing standard. Recommended changes include:

- Revise the annual average standard for PM10 from 30 to 20 $\mu g/m^3$.
- Retain the 24-hour standard for PM10 of 50 $\mu g/m^3$, not to be exceeded.
- Add an annual average standard for PM2.5 of 12 $\mu g/m^3$, given growing evidence from epidemiological and toxicological studies of significant toxicity related to this size fraction of PM.
- Retain the current 24-hour average standard of 25 $\mu g/m^3$ for sulfates.
- Prevent degradation from current ambient air concentrations, measured as PM10 or PM2.5.
- Establish a goal of continued reductions in PM10 and PM2.5 concentrations overtime.

7.1 Particle Dosimetry

For particles to exert any biological effect, they must first come into contact with the target organ tissue: for purposes of this document, the initial target organ of concern is the respiratory tract. In general, particles 10 μm or less in diameter are considered respirable by humans. The depth of penetration into the lung and extent of deposition are determined by a particle's aerodynamic diameter, its ability to attract water (hygroscopicity), electrostatic charge, as well as by host characteristics, including airway structure and geometry, as well as depth, rate and mode of breathing (e.g., nasal vs oronasal). Many inhaled particles are exhaled without depositing in the respiratory tract; the theoretical particle diameter for minimal deposition is about 0.5 μm. In general, for particles with diameters greater than 0.5 and less than 10 μm, increasing size is associated with greater total lung deposition, while for particles with diameter less than 0.5 μm deposition increases inversely with particle size. Soluble particles can be cleared by dissolution into the extracellular fluid lining the airways, with subsequent transport into epithelial or other cells of the respiratory tract, and then into the circulation. Insoluble particles are cleared by more complex mechanisms, as described below.

7.1.1 Deposition

The respiratory tract is often considered to consist of three anatomically and functionally distinct units: (a) the extra-thoracic (ET - from the mouth and nose to the larynx); (b) the tracheo-bronchial (TB – from the larynx through the conducting airways; and (c) alveolar (AL – the gas exchange zone). In general, more serious pollution-related health outcomes are related to effects in the TB and AL regions. The patterns of particle deposition in the respiratory tract do not, however, correspond well to the categories used to classify particles for regulatory purposes (PM10, coarse and fine fractions). Generally, larger particles demonstrate a greater fractional deposition in the ET and upper TB areas, while smaller particles show greater deposition in the deep lung (lower TB and AL). These regional patterns reflect principally the mechanisms of deposition that differentially influence particles by size.

Mechanisms of nonfibrous particle deposition include: (i) gravitational settling, for particles more dense than air; (ii) impaction on the wall of a bronchus or bronchiole, due to inertia maintained when the airstream changes direction at an anatomical bend or bifurcation; (iii) diffusion related to Brownian motion; and (iv) electrostatic attraction, which is generally considered of lesser importance than the other three. Settling and diffusion are more important for particles less than about 3 μm, while inertial impaction generally affects larger particles, particularly in the ET and upper TB area (Foster 1999). For ultrafine particles (with diameters <0.1 μm in diameter), diffusion represents the dominant mode of deposition.

The ET region and especially the nose effectively filter out a large fraction of inhaled particles, mainly those above 1 μm in diameter, but also including ultrafine particles. In general, inertial impaction predominates in the ET region, so increasing particle size and increasing flow rates will tend to increase particle deposition. However, fractional deposition of ultrafine particles (diameters from 0.53 to 0.62 nm, inhaled at flow rates between 5.9 and 22 liters/min) in the nose has also been reported to be very high (in excess of 93%) (Swift et al., 1996).

In the TB and AL areas, increased depth of breathing tends to enhance the deposition of fine particles, while an increased respiratory rate has the opposite effect (Foster et al., 2000). Exercise and increased respiratory rates also tend to result in greater deposition in larger, central airways, and less in the AL region (Foster 2000). Using inert particles 1, 3, and 5 μm in diameter, Kim et al., (1996) showed that, even in healthy adults, there is striking heterogeneity of deposition patterns, with airway surface doses 2 to 16.6 times greater in large airways and up to 4.5 times greater in small airways than in the alveolar region for larger

(3 and 5 μm) particles. A similar, but less pronounced, pattern was also observed for particles of 1 μm diameter. Heterogeneous local particle dose enhancement may also be important among individuals with obstructive lung disease (see below).

Among healthy adults, airway caliber (measured by specific airway resistance) appears to be an important determinant of particle deposition, with a generally inverse relationship between airway diameter and deposition efficiency (Bennett et al., 1996). This may result from the decreased cross-sectional distance that particles have to traverse (by inertial velocity, gravitational settling, or diffusion) before depositing. Women tended to display a greater deposition fraction than men (perhaps because of a smaller respiratory tract anatomy overall). Nevertheless, because men breathed more rapidly than women, they showed a greater deposition of particles per unit time, though the difference was slight when normalized to lung surface area. However, under controlled breathing conditions, women tend to display greater deposition of coarse particles (3 and 5 μm in diameter) throughout the lung, particularly in the ET and TB regions (Kim and Hu 1998). Bennett and colleagues (1996) also found that the deposition fraction of inert fine particles (2 μm) was independent of age among 62 healthy adults (ages 18 – 80), which suggests that among elderly individuals, pre-existing lung disease may be more important than age *per se* with respect to respiratory tract deposition of particles (see below).

Individuals with asthma and chronic obstructive lung disease experience greater fractional deposition of fine particles (diameter of 1 μm) than individuals with healthy, normal lungs, with the degree of particle retention roughly proportionate to the severity of airway obstruction (Kim and Kang 1997). Adult subjects with asthma or COPD showed approximately 1.6- and 2.0-fold greater fractional deposition, respectively, of fine particles than healthy subjects (Kim and Kang 1997). Anderson et al. (1990) showed a similar increase in deposition efficiency of ultrafine particles (0.02 – 0.24 μm) in several individuals with asthma and COPD relative to healthy subjects. This study also included 3 individuals with restrictive lung disease (characterized by lung fibrosis or scarring); these subjects demonstrated ultrafine particle deposition patterns similar to healthy individuals. The enhanced deposition of particles in individuals with chronic obstructive lung disease is likely to have at least four physiological bases: (1) narrowed airways result in increased deposition by inertial impaction; (2) relatively low expiratory flow rates and even airway collapse during expiration allow for longer particle residence time in the lung, favoring deposition of fine and ultrafine particles by diffusion; (3) mucus hypersecretion may cause airflow irregularities that can enhance particle deposition; and (4) uneven ventilation related to airway obstruction may result in deeper particle penetration into those areas of the lung that are still ventilated and functional (Kim and Kang 1997).

In such individuals, one can observe focal hyperdeposition of particles, often in sites of airflow limitation in central airways, even when nominal ambient particle concentrations are relatively low (Foster 2000). Airway hyperresponsiveness, which is one of the hallmarks of asthma but can also occur in otherwise healthy individuals, is likewise associated with enhanced regionalization of deposition to the central airways (Foster 2000). This may exaggerate the patterns of local deposition enhancement observed in healthy individuals (Kim et al., 1996, see discussion above). The work of Kim and Kang (1997) indicates that such dose amplification can occur because, not only do individuals with obstructive lung disease ventilate only a portion of their lungs, but they also will experience increased deposition compared with healthy individuals. Moreover, people with symptomatic asthma or COPD tend to have increased minute ventilation. Assessing these factors together, Kim and Kang (1997) estimate that such individuals may have more than three-fold greater total lung deposition than healthy subjects, with this enhanced deposition concentrated in small areas of the lung.

7.1.2 Clearance

The localization of deposition in the lung will affect the rate, mode, and completeness of clearance. Soluble particles are cleared from the respiratory tract by absorption into extracellular fluids or mucus, then to epithelial cells, from which they can pass into the circulation (Foster 2000). Insoluble ultrafine particles can also be taken up into the respiratory epithelium and have recently been shown to enter the blood of humans within minutes of inhalation, suggesting a potential route for the rapid initiation of systemic particle-related effects (Ferin el al. 1992; Nemmar et al., 2001). However, in general, insoluble particles have been considered to be cleared in two phases: (1) a faster TB phase considered to be more or less complete within 24 - 36 hours, which is effected by mucociliary activity; (2) a more prolonged phase, which can continue for days to months, which is considered to be mediated via engulfment by alveolar macrophages for particles depositing in the deep lung (Foster 2000).

The ciliated airways in the TB region are covered by a thin two-fluid liquid, the upper mucous layer traps particles and transports them up to the throat, propelled by ciliary beating in the lower (sol) layer. Upon reaching the oropharynx, the mucus containing the particles is usually swallowed or expectorated. Carriage on the mucociliary "escalator" is the principal mechanism of the "fast" phase; mucociliary transport rates are generally fastest in the trachea and large bronchi. Some particles may be engulfed by macrophages in the airways, which can then be transported on the mucociliary escalator. However, these processes are not universally successful; some insoluble particles cross into the airway epithelium and enter the lung interstitium (Ferin et al., 1992; Churg and Brauer 1997).

The slow clearance phase has traditionally been considered to affect particles that deposit deep in the lung, beyond the ciliated epithelium. Recent evidence, however, indicates that a substantial fraction of particles depositing in the TB region, particularly the bronchioles, are not cleared for days (Falk et al., 1999; US EPA 1996). Falk et al. (1999) followed the long-term clearance (over a 6-month period) of 6 μm radiolabelled Teflon particles inhaled at 0.5 or 0.05 l/s by human volunteers. The slow inhalation rate facilitates particle deposition that is nearly independent of airway resistance, allowing for greater deposition in the bronchioles. About half the deposited particles remained in the lungs after 24 hr. At the normal inhalation rate (0.5 l/s), 14% of the particles that had not cleared by 24 hr showed a clearance half-time of 3.7 days, while the remaining 86% demonstrated a clearance half-time of 217 days. Of the particles retained at 24 hr after slow inhalation (0.05 l/s), 35% cleared with a half-time of 3.6 days, while the remaining 65% showed a half-time of 170 days (Falk et al., 1999). Thus, for both slow and normal modes of inhalation, there appear to be three phases of clearance: an initial fast phase ($\leq$ 24 hr), an intermediate phase ($t_{½} \approx$ 4 days), and a slow phase ($t_{½} \approx$ 200 days). These investigators assumed that the intermediate phase represented clearance from the bronchiolar region, while the slow phase represented clearance from the AL region.

Alveolar macrophages are the principal clearance vehicle in the AL region. Particle-containing macrophages can make their way to the mucociliary escalator, move to a lymphatic channel within the interstitium to regional lymph nodes, or cross into the circulation, either after passing through the lymph node or possibly by direct entry into the blood across the alveolar capillary endothelium. However, as noted above, clearance processes are not 100% effective: lymph nodes can become storage depots for particles, numerous particles are translocated into the epithelium and interstitium (often within hours of deposition), where they may become aggregated in specific sites around the airways or blood vessels.

Once in the interstitium, particles tend to stay there; clearance is extremely slow, on the order of months to decades. Particle access to the lung interstitium increases as particle size

decreases and particle numbers increase (Ferin et al., 1992). In an examination of autopsy lung tissues of elderly, never-smoking residents of Vancouver (a city with relatively low levels of particulate air pollution; mean PM10 from 1984-1993 = 20 - 25 μg/m^3), Churg and Brauer (1997) found that 96% of particles retained in the lung parenchyma had (calculated) aerodynamic diameters < 2.5 μm, with a geometric mean of 0.41 μ, while coarse and ultrafine particles comprised 4.0 and 4.8%, respectively of the total. Investigating the size and composition of particles retained in the airways among residents of Mexico City as well as Vancouver, Churg and Brauer (2000) found strikingly large numbers of particles (roughly 10^7/g dry lung tissue), with generally increasing quantities proceeding from the mainstem bronchus to the deep lung. The highest concentrations, with particle numbers 25-100 times higher than along the mainstem bronchus, were in the respiratory bronchioles (at the junction between the conducting airways and the alveoli) and at large airway carinas (anatomic bifurcations of the airways). In addition, there were enormous differences (up to several hundred-fold) in particle retention among the study subjects, probably reflecting inter-individual variability in clearance rates.

Exposure to respiratory irritants can stimulate epithelial, sensory neural, and other airway cells to release cytokines and other chemical messengers, and can result in local inflammation, altered epithelial permeability, increased mucus secretion, and bronchoconstriction. Disease states characterized by mucus hypersecretion and disruption of the normal epithelial architecture (e.g., asthma and chronic bronchitis) can produce mucus stasis and adversely affect particle clearance (Foster 2000). As alveolar macrophages engulf substantial quantities of particles, their viability and functional integrity can be adversely affected by PM exposures, due in part to soluble metal-induced oxidative stress (Soukup et al., 2000). Effects on alveolar macrophages may not be limited to fine and ultrafine particles. Kleinman et al., (1995) demonstrated that essential alveolar macrophage functions (phagocytosis and oxidant generation) can be inhibited by coarse particles in re-suspended road dust. In vitro experiments suggest that, in addition to decreasing alveolar macrophage phagocytosis, PM10 exposure appears to reduce resistance to infection with respiratory syncytial virus (Becker and Soukup 1999). Recent work suggests also that ultrafine particle uptake by human alveolar macrophages is common (observed in macrophages obtained from all 14 subjects), and that there may be an inverse relationship between lung function and the extent of ultrafine particle content of alveolar macrophages (Hauser et al., 2001).

Mucociliary clearance can be affected by exposure to acidic aerosols (Schlesinger et al., 1992). In humans, mucociliary clearance has been shown to be depressed following exposures to approximately 100 μg/m^3 sulfuric acid particles for one to two hours (Spektor et al., 1989). In contrast, depression of mucociliary clearance in animals requires concentrations greater than 100 μg/m^3 delivered over several hours or even months (U.S. EPA 1989; Mautz. et al., 1996; Kleinman et al., 1999). Altered mucociliary clearance in humans has the potential to impact incidence of respiratory infection in healthy, as well as compromised, subjects.

7.1.3 Differences between Children and Adults

There are significant anatomic and physiological differences between the developing lungs of children and those of mature adults (Snodgrass, 1992). These include differences in the size and shape of the conducting airways, the number and orientation of physiologically active gas exchange regions and ventilation rates. Though the basic structure of the airways is established in utero, most of the alveoli (≈ 85%) develop in infancy and early childhood. Alveolar multiplication coincides with incorporation of elastin and collagen in the lung, which are responsible for the mature lung's mechanical properties (Lipsett 1995). With growth and development other patterns of anatomical differences emerge: TB airways increase in diameter and length until adulthood, while mean alveolar diameter decreases until about 30

years of age (Thurlbeck 1991). Repeated episodes of PM-related injury and inflammation may therefore have long-term consequences on the lung's functional abilities (see section 7.6, below).

Because of differences in anatomy, activity, and ventilation patterns, children are likely to inhale and retain larger quantities of pollutants per unit body weight than adults (Adams 1993). Phalen et al. (1985) developed a model incorporating airway dimensions measured in lung casts of people (aged 11 days to 21 years) to predict that particle deposition efficiency would be inversely related to body size, which would tend to accentuate differences in exposure related to activity and ventilation patterns. Corroborative evidence for this was provided by Oldham et al. (1997), who found that in models of the proximal TB airways (i.e., the trachea and the first two bronchial bifurcations) of 4- and 7-year-old children and an adult, deposition efficiencies for radiolabeled particles 1.2, 4.5, 9.7 and 15.4 μm in median aerodynamic diameter were greater in the child models in almost all cases. As expected, particle deposition efficiency increased markedly with increasing particle size in this model system; for instance, in the model of the four-year-old child, the deposition efficiency increased from 0.3% to 10.7% when the smallest and largest particle sizes were used, respectively.

Inhalation experiments comparing particle deposition patterns in children and adults have produced somewhat inconsistent results. Schiller-Scotland et al. (1994) reported greater fractional deposition in healthy children, aged 3 – 14 years, compared with adults, when breathing 1, 2 or 3 μm particles spontaneously through a mouthpiece. The differences were greater with the larger particles. However, as noted by the authors, these children were breathing more deeply than expected, which is a common tendency when breathing through a mouthpiece. This propensity may result in greater time-dependent deposition of fine particles (by sedimentation and diffusion). Schiller-Scotland et al. (1994) also noted that, among the older children (mean age = 10.9 years) who were capable of controlled breathing (in time with a metronome), that particle deposition was inversely related to body height, so that the shorter children demonstrated greater fractional deposition (for 1 and 2 μm particles, the only categories analyzed in this manner). In contrast, Bennett et al. (1998) found no significant differences between children (7 – 14 yr), adolescents (14 to 18 yr), and young adults (19 – 35 yr) in deposition (measured as deposition fraction or rate) of 2 μm (mass median aerodynamic diameter - MMAD) particles during spontaneous breathing at rest. Unlike the study by Schiller-Scotland et al. (1994), this investigation tailored the participants' mouthpiece breathing patterns to those measured during unencumbered breathing (using resistance impedance plethysmography), in order to control for the tendency to breathe more deeply through a mouthpiece. Another difference between the study by Bennett et al. (1998) and that by Schiller-Scotland et al. (1994) is that the former did not include very young children, who would have had difficulty in mimicking their normal breathing patterns while using a mouthpiece. However, Schiller Scotland et al. (1994) found that older children (mean age = 10.9 years) as well as the younger ones (mean age = 5.3 years) also showed increased fractional particle deposition relative to adults.

Children demonstrate lower absolute minute ventilation at rest than adults, despite having higher breathing rates. Relative to lung volume, however, children demonstrate a higher minute ventilation than adults. Thus, Bennett et al. (1998) noted that children tended to have a somewhat greater normalized deposition rate (by about 35%) than the combined group of adolescents and adults, suggesting that children at rest would receive higher doses of particles per unit of lung surface area than adults. This tendency might be additionally enhanced by activity patterns, as children spend more time than adults in activities requiring elevated ventilation rates. However, it is unknown whether flow-dependent deposition

mechanisms operative at higher ventilation rates in children would offset the decreases that would occur in time-dependent mechanisms (sedimentation and diffusion). If this offset does occur, then particle deposition would likely be shifted more towards the larger, more central airways, which would tend to increase the dose per surface area in children versus adults (Bennett et al., 1998).

7.2 Overview: Epidemiological Studies of Airborne Particulate Matter

Particulate matter (PM) is a heterogeneous, complex mixture of liquid and solid particle sizes and chemicals; thus, it has been difficult to conduct animal or human clinical studies using mixtures found in ambient air. Until the recent development of ambient air particle concentrators, toxicological and controlled human experiments involving PM have generally used simple model particles (e.g., sulfuric acid) or mixtures taken from a single source (e.g., diesel exhaust or residual oil fly ash). In contrast, some health effects of gaseous pollutants can be studied directly using controlled concentrations in chamber experiments. Therefore, most of the health evidence on PM has been derived from observational epidemiological studies of human populations in a variety of geographic (principally urban) locations. Most of the studies have examined short-term or acute consequences (i.e., those occurring on the same day as or within a few days of the exposures of interest) of PM exposure on health, including both mortality and morbidity. Studies of the acute effects of PM exposure typically involve daily time-series observations collected over several months or years. The studies often examine whether daily counts of mortality or cause-specific hospitalizations are correlated with daily concentrations of PM, after controlling for effects of other covariates and potential confounders. Such factors may include temporal and meteorological variables, e.g., day-of-the-week, extremes in temperature, humidity or dewpoint, co-pollutants, and longer-term trends represented by seasonal changes or population growth. Well designed time-series studies can have several methodological strengths, including: (1) a large sample size (sometimes up to 4 to 8 years of daily data, while other times a panel of approximately 100 people observed daily over a 3 month period), conferring substantial statistical power to detect effects; (2) implicit incorporation of a wide range of population demographics, baseline health characteristics, and human behaviors, enhancing the generalizability of the results; (3) real-world exposures, avoiding the need to extrapolate to lower concentrations or across species; (4) the ability to examine effects in potentially sensitive individuals, children and infants; and (5) a limited number of covariates or potential confounders, notably other pollutants and weather factors. Limitations of or potential uncertainties associated with time-series studies include: (1) difficulty in determining actual pollutant concentrations to which people are exposed; (2) the potential for misclassification of exposure; (3) the potential for omission of important explanatory factors or inappropriate control of potential confounding factors; (4) difficulty in measuring or observing all potential health effects; (5) covariation among pollutants making it difficult to attribute an effect to a single pollutant. Moreover, the average daily PM10 concentration in a given location will be similar to the annual average PM10 concentration. While relationships between health outcomes and acute exposures can still be identified through time-series analysis, it is difficult to determine the effect of a single 24-hour exposure independent of the influence of low-level chronic exposures. This potential difficulty represents one aspect of the exposure misclassification. Nevertheless, the epidemiological studies of PM provide a major body of evidence regarding the associated health effects, and serve as a basis for many of the conclusions and recommendations that follow.

7.3 Daily Exposure – Mortality

Over the past two decades, several dozen time-series studies spanning five continents have demonstrated associations between daily counts of mortality and daily or multi-day changes in the concentrations of several common air pollutants. Among these pollutants, various particulate matter metrics – including PM10 (particulate matter with a median aerodynamic diameter equal to or less than 10 microns), PM2.5 (particulate matter with a median aerodynamic diameter equal to or less than 2.5 microns), black smoke, and sulfates – appear to show the most consistent associations with mortality, although some associations have also been reported for ozone, sulfur dioxide, carbon monoxide, and nitrogen dioxide.

Time-series studies examine daily changes in air pollution, typically based on a single 24-hour average, in relation to daily counts of mortality. The analysis typically uses multivariate regression models that control for potential confounding factors other than pollution that may vary over time and may also be associated with mortality. Such factors include day of the week, season, weather, time, and co-pollutants. For example, there is evidence that meteorological factors, such as extremes in temperature and humidity, are associated with mortality. Similarly, there have been consistent observations of cause-specific mortality patterns related to the day of the week. Failure to control for such effects could bias the estimated effects of air pollution. All of the mortality studies associated with short-term exposure reviewed below incorporated statistical control for the effects of weather. In addition, two studies (Samet et al., 1998; Pope and Kalkstein, 1996) involved very detailed modeling of weather patterns with the aid of a meteorologist. These studies found that the estimated effects of PM were not affected by the more complex consideration of weather factors. Likewise, population increases over time must be taken into account since they could, by themselves, explain some of the increases in daily mortality. In addition, in cities with temperate climates throughout the world, colder winter seasons are associated with more respiratory disease and mortality. Again, failure to adjust for seasonal patterns in mortality could lead to a false attribution of these effects to air pollution.

Most of the air pollution-mortality studies published over the last decade employ statistical techniques that control for these potentially confounding influences. In particular, recent, higher-quality studies are characterized by: (1) use of Poisson regression models, since mortality is a rare event and can be described by a Poisson distribution; (2) three or more years of daily data in a given city or metropolitan area; (3) examination of the effect of day-of-the-week and daily changes in the weather; and (4) use of locally weighted smoothing (loess). The latter is a technique that can account for both time trend and seasonal patterns (due to variations in weather and population susceptibility) in daily mortality data. The loess smoothing technique can accommodate nonlinear and nonmonotonic patterns between time and other factors and the health outcome, offering a flexible nonparametric modeling tool. Including a smoothed variable in the model does not explain the underlying reason for the pattern over time, but controls for it statistically, allowing one to observe the relationship between daily mortality and environmental factors after the underlying trend in daily mortality is controlled for. In addition, adding a locally weighted smooth of time diminishes short-term fluctuations in the data, thereby helping to reduce the degree of serial correlation. Serial correlation exists when the errors of the regression model are related over time, producing biased estimates of the variance of the explanatory variable coefficients.

With increasing statistical sophistication, these studies have shown that either one-day or multi-day PM average concentrations are associated with both total and cardiopulmonary mortality. However, while acute exposures appear to exert an independent effect on mortality the influence of a single 24-hour exposure at a concentration relevant to the PM standards, absent any other exposure to PM, has not been (and probably cannot be) determined

epidemiologically. Our review focuses primarily on those studies that used PM10 or PM2.5 as the exposure metric. Other measures of PM include black smoke (BS), coefficient of haze (COH), and sulfates.

7.3.1 General Results

There are now many studies linking short-term (i.e., daily) changes in PM10 with premature mortality. This includes not only studies from throughout the U.S., including several from California, but also those from a diverse group of cities throughout the world: such as Santiago, Chile (Ostro et al., 1996), Mexico City (Castillejos et al., 2000), Sao Paulo, Brazil (Saldiva et al., 1995), Amsterdam (Verhoeff et al., 1996), Bangkok (Ostro et al., 1999a) and Sydney (Morgan et al., 1998). Such cities span a wide range of environmental and population characteristics, including temperature – air pollution relationships, housing stock, transportation systems, industrial emissions, population age distributions, typical activity patterns, and baseline health conditions. Meta-analyses of earlier mortality studies suggest that, after converting the alternative measures of particulate matter used in the original studies to an equivalent PM10 concentration, the effects on mortality are fairly consistent (Ostro, 1993; Dockery and Pope, 1994; Schwartz, 1994a). Specifically, the mean estimated change in daily mortality associated with a one-day 10 $\mu g/m^3$ change in PM10 implied by these studies is approximately 0.8 percent, with a range of 0.5 percent to 1.6 percent. Since these meta-analyses were published, many more studies of acute exposure-mortality have been completed. All include control for weather and other potential confounding factors and most use sophisticated smoothing techniques as well. Table 7.1 summarizes the acute exposure mortality studies that have directly measured PM10. The table provides information for single-pollutant models of all-cause mortality, using the lags demonstrating the highest associations with mortality, based on t-statistics.

Among the first of the multi-city studies on mortality, Schwartz et al. (1996) examined data from the Harvard Six-City studies. This database included monitors that were specifically sited to support ongoing epidemiological studies and be representative of local population exposures. A consistent association was reported between daily mortality and daily exposures to both PM10 and PM2.5. The mean concentrations of PM10 among the six cities ranged from 18 to 47 $\mu g/m^3$ (overall mean of 30 $\mu g/m^3$) with a joint effect estimate indicating a 0.8% (95%CI = 0.5 – 1.1) increase in daily total mortality per 10 $\mu g/m^3$ of PM10.

Samet et al. (2000a) applied a wide range of statistical tools and sensitivity analyses to a database consisting of the 88 largest cities in the United States, while Samet et al. (2000b) focused on the 20 largest cities. For both of these studies, the combined effect of all of the cities indicated an association consistently within but near the lower end (approximately 0.5% per 10 $\mu g/m^3$ of PM10) of the range reported by earlier researchers. Among these cities, mean PM10 ranged from 24 to 46 $\mu g/m^3$. The authors examined pollution and sociodemographic factors that might modify the estimated effects of PM10. They reported no association between the effect estimates for each of the cities and the mean level of PM or other pollutants (ozone, nitrogen dioxide, sulfur dioxide or carbon monoxide) in the city. This suggests a constant slope or effect per $\mu g/m^3$ of PM regardless of the average concentration of PM or other pollutants. In addition, city-wide estimates of sociodemographics such as median income, percent unemployed, and percent below poverty level did not modify the estimated effect of PM. However, there may have been lack of statistical power to detect an effect, if one existed.

Samet et al. (2000a) indicated that their estimates may be at the lower end of the range because their database included a wide range of cities and incorporated findings in some cities where no effects were observed. There may be other explanations for the lower effects,

however. For example, the studies only considered lags (or delayed effects) of zero, one day, or an average of these two days, although other studies have reported greater effects with longer lags or multi-day moving averages. Since many of the cities in the study collected PM10 data on an every-sixth-day basis, these averaging times could not be examined. Another possible reason for the lower effect estimates in the study by Samet et al. (2000a) relates to the number of covariates used in the regression model. Besides PM10, day of week, and a smooth of time using 7 degrees of freedom (or cycles of about 7 weeks), two variables were included for temperature and two for dewpoint (same day and an average of the three previous days). Most previous mortality studies used fewer controls for weather factors. To the extent that PM may be causally related to mortality and correlated as well with these meteorological variables, these multiple statistical controls could result in an underestimate of the effects of PM. Thurston and Ito (2001) demonstrated that the modeling of weather factors had a significant impact on the estimated effect of ozone, and postulated that it could impact the estimated effects of secondary aerosols, as well.

The largest and most significant regional effects were found for the Northeast U.S. and for Southern California, with marked heterogeneity in the PM-mortality relationships from region to region. The regional heterogeneity may have resulted from differences in: (1) the particle composition and size distributions; (2) the underlying distributions of age, chronic disease, and other determinants of susceptibility among the local populations, including behaviors and activity patterns, and exposures; or (3) the density of pollutant monitors and relative exposure measurement errors. Moreover, the application of a similar statistical model to all 90 cities may have contributed to the inter-city and inter-regional variability observed by these researchers. Similar smoothers of time and temperature were used throughout the country, despite the diversity of climate, PM sources, and population characteristics. By not tailoring the model to each locale, they may have had varying degrees of "goodness-of-fit" of the models to the mortality patterns in the individual cities, which might either exaggerate or underestimate the magnitude of the associations between ambient PM and daily mortality in different locations. In the Samet et al. (2000a) analysis, the averaged effect for the six California counties studied (Los Angeles, San Diego, Orange, Santa Clara, San Bernardino and Alameda) was 0.9% per 10 $\mu g/m^3$ (with a range of 0.3% to 2.0%) versus 0.5% for all 90 cities together. The same data set was used to address issues relating to potential exposure measurement error bias and confounding by co-pollutants. They found that measurement error would likely underestimate the effect of PM (Zeger et al., 2000) and that co-pollutants such as ozone, nitrogen dioxide, sulfur dioxide, and carbon monoxide did not confound the estimated effect of PM (Samet et al., 2000a).

In studies of 10 U.S. cities, Schwartz (2000a) examined the effects of PM10 for all age groups, while Schwartz (2000b) considered only individuals above age 65. For the group of all ages, a 10 $\mu g/m^3$ change in PM10 (average of lag 0 and lag 1 days) was associated with a 0.7% increase in daily mortality. For the elderly age group, the same change in PM10 was associated with a 1.1% increase in mortality. For these cities, the arithmetic mean of PM10 ranged from 27 to 41 $\mu g/m^3$.

In another multi-city study, Burnett et al. (2000) analyzed mortality data for 1986- 1996 from the eight largest Canadian cities. This study found that both PM10 and PM2.5 were associated with daily mortality. For PM10, a 10 $\mu g/m^3$ increase was associated with a 0.7% (95%CI = 0.2 – 1.2) increase in daily mortality, with a mean PM10 concentration of 26 $\mu g/m^3$. Moolgavkar (2000a) examined the association between air pollution and mortality in three large U.S. counties: Cook (including Chicago), Maricopa (including Phoenix), and Los Angeles, for 1987 through 1995. For the latter two counties, only every sixth day measures of PM10 were available, unlike most of the other studies which had daily data (except Samet et

al., 2000a, b). PM10 was significantly associated with mortality in all three counties but with a lower effect estimate (approximately 0.2 to 0.4% per 10 $\mu g/m^3$) than found in most other studies. In addition, the author concluded that it was difficult to assign the effect to any single pollutant because of the high correlation among pollutant measurements.

Another multi-city study involved 29 European cities that measured PM10 (although in some of the cities PM10 was estimated from observations collected from a subset of days using collocated TSP or Black Smoke monitors) (Katsouyanni et al., 2001). Using a methodology similar to the U.S. studies cited above, an association between daily mortality and PM10 was reported with an overall effect estimate of 0.6% per 10 $\mu g/m^3$. The study reports heterogeneity in the effect estimates, which was likely due to real differences in PM sources and exposures among the cities. In this regard, cities that had higher concentrations of nitrogen dioxide, indicating the likelihood of a greater contribution of ambient pollution from mobile sources, especially diesel, demonstrated greater PM10-associated effects. For example, for cities in the lowest quartile for nitrogen dioxide, the estimated PM10 effect was 0.2% per 10 $\mu g/m^3$, while for cities in the highest quartile for nitrogen dioxide the effect estimate was 0.8% per 10 $\mu g/m^3$.

Single city studies reporting an effect on mortality from acute exposure to PM have been conducted in over 100 cities. Those studies that specifically use PM10 (as opposed to Black Smoke, Coefficient of Haze (COH), nephelometry data or other measures of PM) as their exposure metric are summarized in Table 7.1, which displays the estimated effect and ambient concentration of PM10 for each city. As in the studies conducted in the early 1990s, these studies indicate a mortality effect of around 1% per 10 $\mu g/m^3$ of PM10. Taken together and combined with the evidence of morbidity effects described below, these studies provide compelling evidence of a significant impact of PM on mortality. Although the relative risk per unit is low, the large number of people exposed suggests the existence of a major impact on public health.

Many of the above studies reported that lags in exposure to PM10 of one to four days exhibited stronger associations with mortality than did contemporaneous exposure to PM10. In addition, cumulative exposure of three or five days, when tested, often had stronger associations than single-day lags. Recent analysis has demonstrated that the effect estimates increase when a longer-term average of exposure is used. For example, Schwartz (2000b) examined mortality for those above age 65 in 10 U.S. cities. A regression model that allowed for an air pollution effect to persist over several days using a distributed lag was incorporated, resulting in a doubling of the relative risk, to approximately 2% per 10 $\mu g/m^3$ of PM10.

In a separate study restricted to out-of-hospital deaths (i.e., excluding those due to homicide or trauma), the effect size increased four-fold (Schwartz, 2001a). Schwartz (1994b) had previously found a much greater likelihood of deaths occurring outside of hospitals or clinics on days with high versus low concentrations of PM. These findings suggest that particulate air pollution may have a greater impact among individuals who were not in the hospital when exposed and who were not admitted to the hospital before expiring. Sudden death may therefore be an important factor in air pollution-related mortality, which suggests that the average impact on loss of life is likely to be more than just a few days, since it need not include only those already chronically ill and hospitalized.

Analytical results of these studies also indicate that the associations between PM and mortality are not significantly confounded by weather patterns, longer-term seasonality, or day of week. This evidence is provided by careful modeling and controlling for these factors in the individual studies, as well as by the heterogeneous nature of the cities examined. Specifically, consistent evidence of an effect of PM has been observed in cities in both cold (e.g., Detroit

and Montreal) and warm (e.g., Mexico City and Bangkok) climates, in some cities where PM peaks in the summer (Steubenville, Philadelphia) and in others with peaks in winter (e.g., Utah Valley) or spring (Helsinki), and in cities with substantial seasonal changes in mortality (e.g., Chicago) and others with little seasonality (e.g., Coachella Valley, Birmingham, Bangkok). These factors are carefully modeled and controlled for in the studies, and the mortality results are consistent throughout, thereby providing compelling evidence of an effect. Further, factors such as smoking, exposure to secondhand smoke or occupational irritants, and personal characteristics are not confounders in these studies since they do not vary with air pollution on a daily basis.

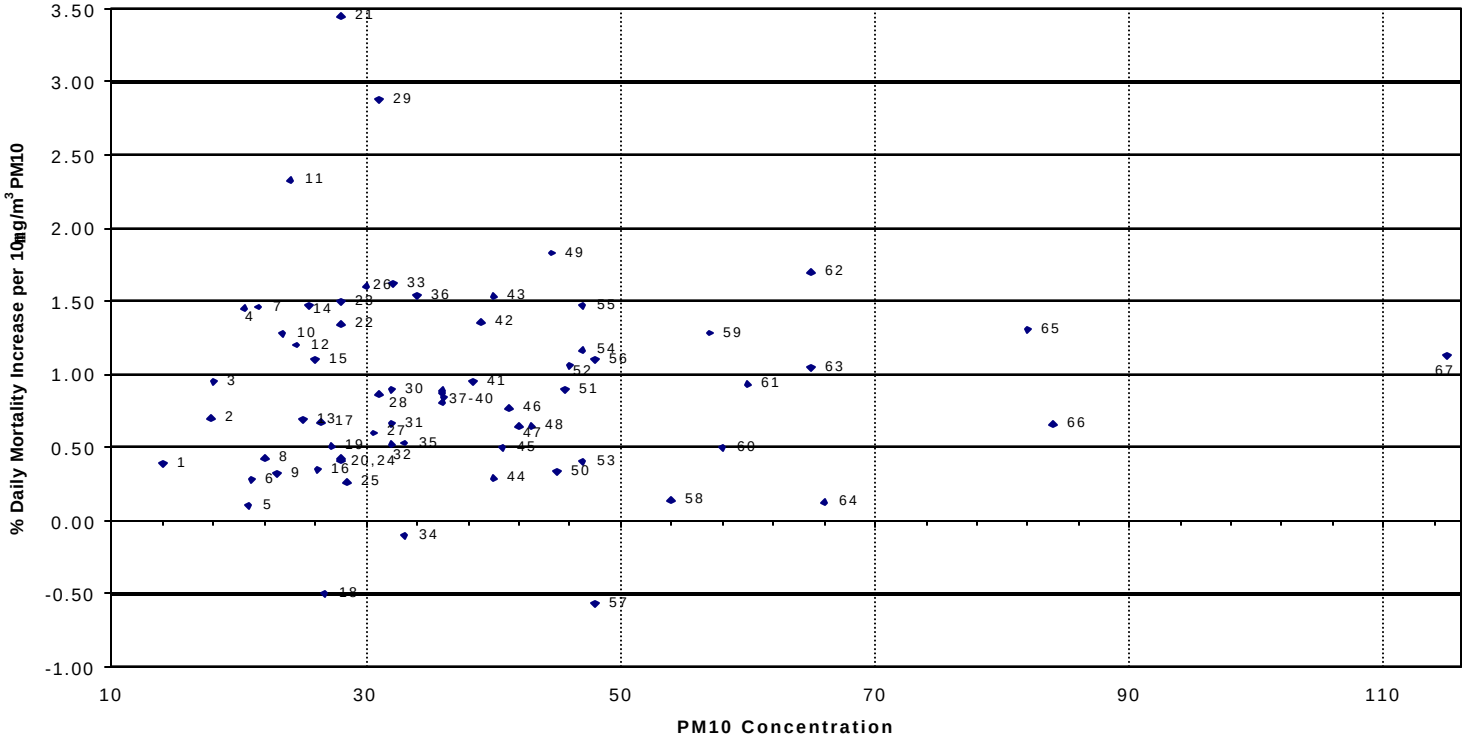
A related issue is whether there is independent evidence of an effect of PM, or whether confounding by co-pollutants makes it impossible to implicate PM as a pollutant of concern. One method for examining such potential confounding involves including multiple pollutants in the explanatory regression model. While this method can help rule out confounding effects if the effect of PM10 is unchanged when other pollutants are included in the model (assuming non-differential measurement error), the reverse is not true. If the estimated effect of PM10 is altered after inclusion of other pollutants, this may be a predictable result of statistical collinearity. It is well established that regression estimates can vary widely with the inclusion/exclusion of highly correlated covariates. It may also be the result of differential pollutant measurement errors or monitor performance.

Despite these potential limitations, there is substantial evidence from the available literature that PM effects are, in general, not substantially impacted by co-pollutants. Samet et al. (2000a) provide a recent and comprehensive test of this theory using the data set consisting of 90 U.S. cites, as described earlier. The authors sequentially tested the estimated effect of PM10 after gaseous pollutants (ozone, nitrogen dioxide, sulfur dioxide, and carbon monoxide) were each added to the regression model. The authors report minimal change in the estimated PM10 coefficient after these inclusions. Similar results have been reported in most studies that have examined PM10 and mortality, with few exceptions (e.g., Moolgavkar, 2000). In a different approach to the issue, Schwartz (2000a) examined the sensitivity of the PM10 coefficient to different amounts of co-pollutant covariation among 10 U.S. cities. Theoretically, if the PM10 effect were really a result of confounding by another pollutant, the estimated PM10 effect per $\mu g/m^3$ would be greater in those cities where PM10 was highly correlated with other pollutants, indicating that PM10 was taking on some of the explanatory power of the "true" causal co-pollutant. The author did not find any evidence, however, consistent with this hypothesis, suggesting that confounding of the effects of PM10 by other pollutants was unlikely. Similarly, in the study of 29 European cities, Katsouyanni et al. (2001) report no effect modification or confounding associated with either ozone or sulfur dioxide. PM effects were higher in cities with higher concentrations of nitrogen dioxide, but the effects of PM were not attenuated.

We have attempted to provide a context for both the average ambient concentrations and the statistical level of uncertainty in these studies. Figure 7.1 and Table 7.1 summarize the estimated effect levels and the associated average concentrations for the available studies that used PM10. (Unpublished data for individual city results within multi-city studies were graciously supplied by the authors). This obviates the need to adjust from some other PM measure such as black smoke to PM10, and thereby reduces one source of uncertainty. The figure indicates that many studies show associations between daily exposure to PM10 and mortality that have average PM10 concentrations in the range of 20 to 30 $\mu g/m^3$. However, all of the published studies at the lower end of the range have been conducted outside of California, and several are from outside the U.S. The cities are sorted by PM10 concentration in Table 7.1 and show, for example, that the 10 lowest concentrations occur in Stockholm,

Portage (Wisconsin), Sydney, Ottawa, Edinburgh, Vancouver, Paris, Helsinki, and Edmonton. Thus, extrapolation from studies involving the lowest concentrations of PM10 to California may involve additional uncertainties, since these cities may be very different from California cities. Factors that may affect the PM- mortality relationships, including sources of PM, different distributions of PM size and chemical compositions, time spent outdoors, proximity to the roadways, climate, population age distribution and health status, smoking characteristics, and use of medical care may all be different. These factors may lead to either stronger or weaker effects in California, but in general makes the extrapolation less certain.

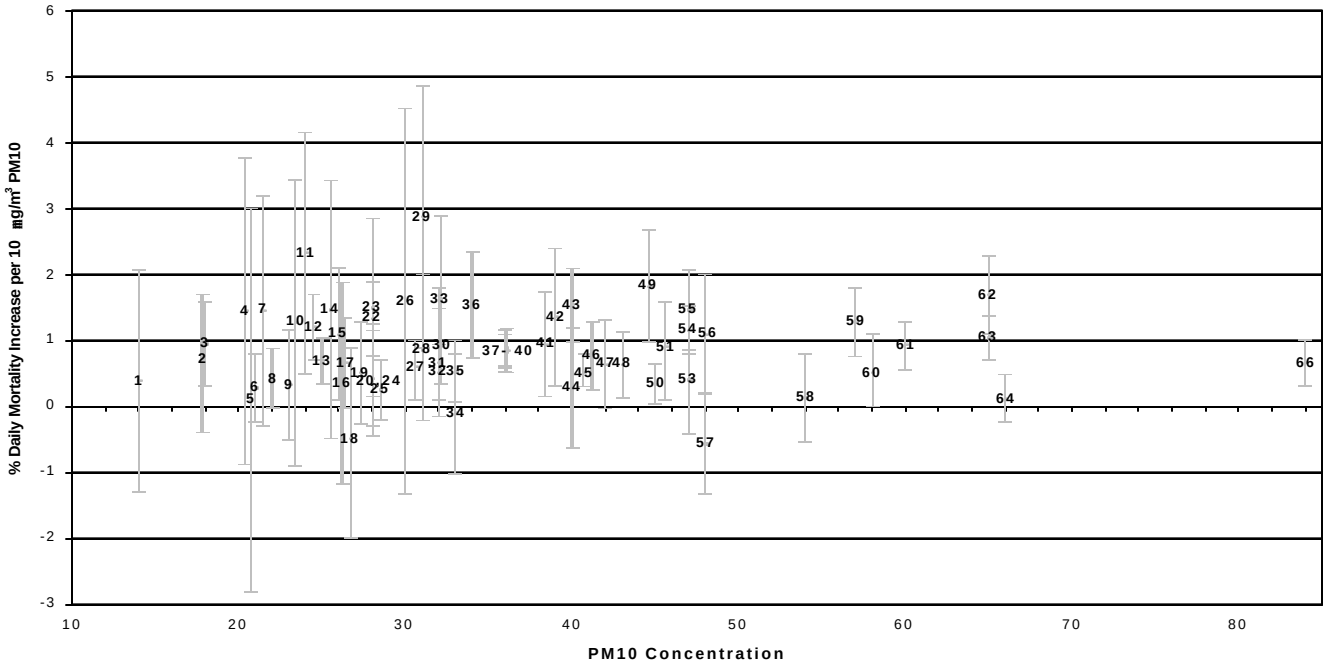
Figure 7.1 Daily Mortality Estimates and PM10 Concentration



Note: Both median and mean are used to indicate average study concentration. Number in the figure refers to city identifier, see Table 7.1 for study details.

Figure 7.2 demonstrates that the studies themselves may involve greater uncertainty at lower mean PM10 ambient concentrations. As the average PM10 level decreases, the confidence intervals of the estimated effect on mortality tend to increase. The associated t-statistic (which equals the regression coefficient divided by the standard error of the estimate) is a unit-free measure of the association in each of the regressions. The larger the t-statistic, the stronger the association and the smaller the 95% confidence interval associated with the estimated effect. Therefore, Figure 7.2 also indicates that at lower ambient concentrations, the t-statistic tends to be lower as well. This simple figure, however, does not account for other possible factors that may be confounding this relationship. For example, it may be the case that studies conducted in generally less polluted cities have other factors that affect the association, such as weather, particle composition, or housing stock (i.e., with different levels of "tightness" and infiltration rates). Therefore, Figure 7.2 can only be considered suggestive regarding the greater degree of uncertainty at lower concentrations. It should be noted that many studies have found statistically significant associations between PM10 and mortality at low ambient concentrations and that analyses explicitly conducted to determine thresholds have failed to detect any (see Section 7.3.5 below). Therefore, Figure 7.2 should not be construed as demonstrating a threshold of a level of zero risk. It also should be noted that the large (n = 88) multi-city study of short-term exposure and mortality by Samet et al. (2000a) found that although the magnitude of the estimated mortality effect varied across all of cities (and tended to be associated with PM within each city), the effect estimate was independent of the mean PM10 in any given city. Thus, cities with higher average concentrations of PM10 tended to have the same general effect per microgram of PM10 as cities with lower averages.

Figure 7.2 Uncertainty in Daily All-Age Mortality Studies versus Study



Note: Bars represent 95% CI of estimated PM10 effect; number in the figure refers to city identifier. The city identifier is placed at the point estimate location. Santiago, Chile does not appear in this graph. See Table 7.1 for study details.

The t-statistic associated with the estimated coefficient of PM10 will be affected by both the strength of the association between PM10 and mortality, and the number of observations used in the regression model. Theoretically, the t-statistic should increase with the square root of the number of observations. In order to control for this factor and still determine whether the concentrations of PM10 were associated with greater uncertainty, we conducted a simple statistical analysis of the 62 single-city studies for which we had complete data for all-cause mortality for all age groups together (see Table 7.1 for details of the studies). Only all-age, all-cause mortality results are included, using the lag with the highest association with mortality, based on the t-statistic. In the analysis, we used ordinary least squares multiple regression to explain variations in the t-statistic as a function of both the number of study observations (days) and the average concentration of PM10. We also used locally weighted smoothing analysis (Cleveland and Devlin, 1988) to examine the shape of the possible associations. Both the concentration of PM and the square root of the number of days in the study appear to have linear associations with the t-statistic, Specifically, we found the following relation:

$$\text{Tstat} = -0.39 + 0.025\ \text{SRN} + 0.0528\ \text{PM}$$

$$(0.019) \qquad (0.0129)$$

$$p = 0.18 \qquad p < 0.0001$$

$$R^2 = 0.25$$

where Tstat = t-statistic of the association between PM10 and mortality,

SRN = square root of the number of days of the study,

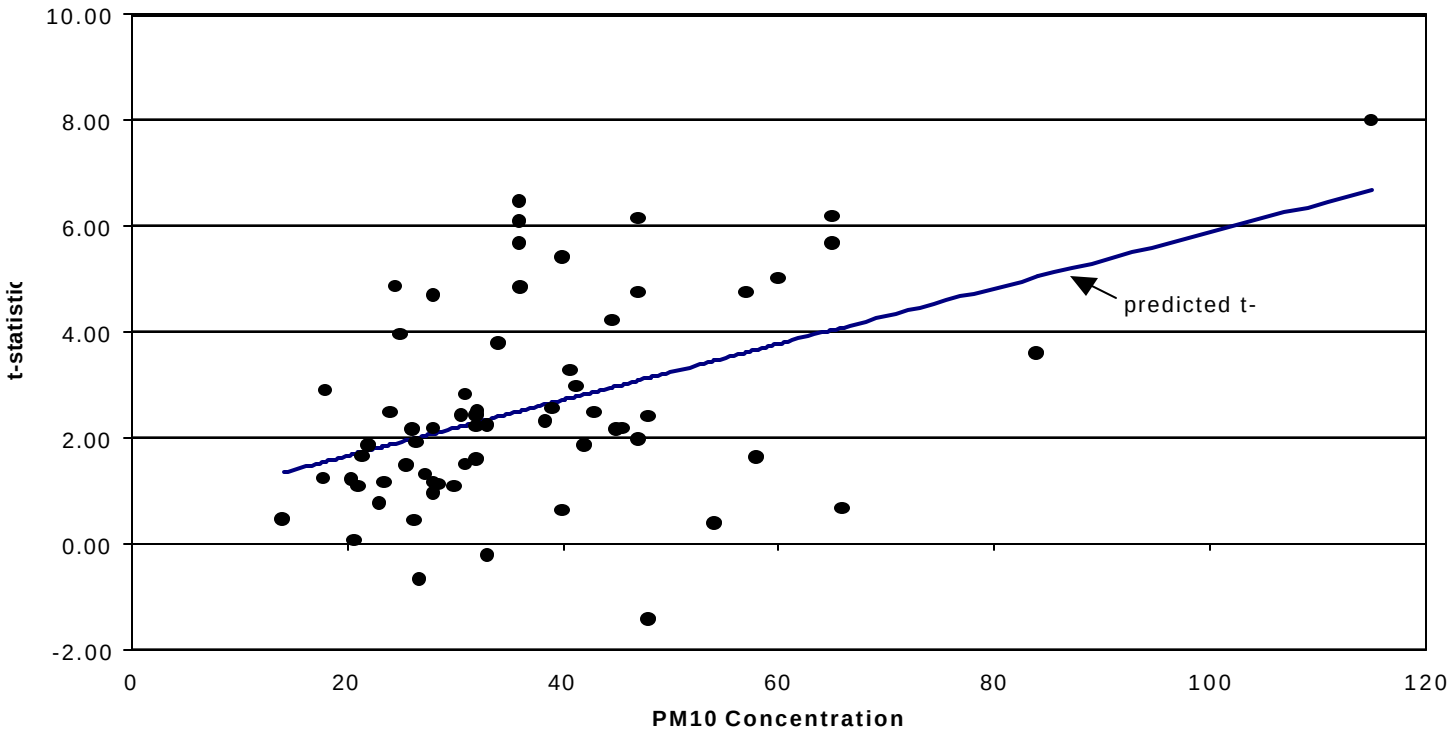
PM = average study concentration of PM10, and

the standard error is in parentheses.

The estimated coefficients indicate that uncertainty (the inverse of the t-statistic) decreases with increasing sample size and PM concentration. The coefficient of N had a p-value of 0.18, while the coefficient of PM had a p-value < 0.0001, indicating that the mean PM concentration may be an important determinant of the level of uncertainty in these studies. About 25% of the variation in the dependent variable Tstat was explained by the two terms. The lack of statistical significance of the study days likely indicates some residual confounding by other factors related to characteristics of the city. Similarly, the statistical significance of the study PM10 concentration will also be impacted by covariates that are unmeasured in the analysis. However, the high precision of that estimate suggests that PM10 concentrations would still be an important predictor even with the inclusion of other covariates in the model.

Figure 7.3 displays a plot of the linear fit for the predicted value of Tstat versus average PM10 concentration, after controlling for number of observations. While this simplistic analysis does not control for a wide range of other factors that may affect the strength of the association, it does suggest greater uncertainty at lower concentrations. The plot also indicates that there are two outliers: the observations associated with the highest and lowest t-statistics. Therefore, as a sensitivity analysis, the model was rerun after deleting these two points. The resulting model produced a slightly lower coefficient for PM10 of 0.046 (s.e. = 0.015, $p < 0.01$), a higher coefficient for SRN of 0.031 (s.e. = 0.018, $p < 0.10$) with an $R^2 = 0.20$. Thus, both the number of observations and the study average concentration of PM10 were associated with the t-statistic of the estimated effect of pollution.

Figure 7.3 Comparison of t-statistic from Daily All-Age Mortality Studies and Study PM10 Average



Note: Regression fit after controlling for number of observations in study.

7.3.2 Effects by Size Cuts: Fine and Coarse Particles

In the last several years, several daily exposure-mortality studies have examined associations using different particle cut sizes, especially fine (PM2.5) and coarse (PM10 – PM2.5) (abbreviated below as FP and CP, respectively). The ability of these epidemiological studies to differentiate between the effects of different measures of PM size cuts, however, is limited by two factors. First, PM metrics in a given region are often highly correlated. For example, in many urban areas, FP and PM10 are highly correlated ($r > 0.7$) on a daily basis. On the other hand, in areas where crustal PM predominates, daily concentrations of PM10 are correlated with CP. The second factor that limits the interpretation of the epidemiological studies is the relative degree of exposure measurement error. Since FP tends to be more uniformly spatially distributed than CP, it is likely that a fixed-site monitor will be less precise in measuring the latter. Since misclassification of exposures would normally result in biasing the estimated effect downwards, the relative difference in measurement error could lead to relatively lower (and less certain) effect estimates for CP.

Earlier studies of FP used measures of components of FP, such as sulfates (Bates and Sizto, 1987), or used estimates of FP based on airport visibility (Ostro, 1995). Schwartz et al. (1996) was among the first studies using actual measures of FP in the Harvard Six-Cities data set, and then determining CP using the difference between PM10 and FP. Based on both the individual-city analyses and a meta-analysis of all six cities, an association was demonstrated between daily mortality and FP, but not CP. An effect of CP was observed in only one of the six eastern and mid-western cities (Steubenville, Ohio) included in the database. In this study, the mean FP among the cities ranged from 11 to 30 $\mu g/m^3$ with a mean of 18 $\mu g/m^3$, while CP ranged from 7 to 16 $\mu g/m^3$, with a mean of 11.5 $\mu g/m^3$. These findings were validated in an independent replication of the six-Cities data by Klemm et al. (2000).

Among more recent studies (summarized in Table 7.2 and Figure 7.4) examining the relative impacts of coarse and fine particles, however, the results have been mixed. The estimated effects of PM appear to depend on: (1) the cities being studied; (2) the lags in exposure used in the statistical models; (3) the mortality endpoint(s) under study (i.e., all-cause versus cardiovascular or respiratory); and (4) the season(s) under study. In some cities, only a FP effect is found. In other cities, both FP and CP are associated with mortality, while in a third set of cities, an association is only found for CP. Table 7.2 provides a summary of these findings. For example, support for a dominant FP effect is provided by the Fairley (1999) study of Santa Clara County, California. In this study, PM10 (mean = 34) and FP (mean = 13) were associated with all-cause daily mortality, whereas no effect was observed for CP. When cardiovascular mortality was examined in relation to the three different PM cut sizes, associations were found for only PM10. A similar result was reported for all-cause mortality in a study of eight Canadian cities (Burnett et al., 2000). The effect of FP on mortality was stronger than that of CP, although the latter did demonstrate a positive, though weaker, association with mortality.

In contrast, results from Coachella Valley, CA (which includes Palm Springs), Detroit and Mexico City suggest effects of CP greater than those of FP. In PM data from Coachella Valley, Ostro et al. (2000) found very high correlations between CP and PM10 ($R \sim 0.95$) with the ratio of CP/PM10 of approximately 0.60. This is the reverse of most urban areas, particularly in the eastern part of the U.S., where FP is more highly correlated with PM10 and the FP/PM10 ratio is typically between 0.55 and 0.75 (U.S. EPA, 1996). Using 2.5 years of data of PM10 and FP, both CP and PM10, but not FP, were associated with cardiovascular mortality (Ostro et al, 2000). For all-cause mortality, no associations were found for the alternative measures of PM, with the exception of a 4-day lag in FP, which was the only single-day lag demonstrating a positive association. In a previous study conducted in the

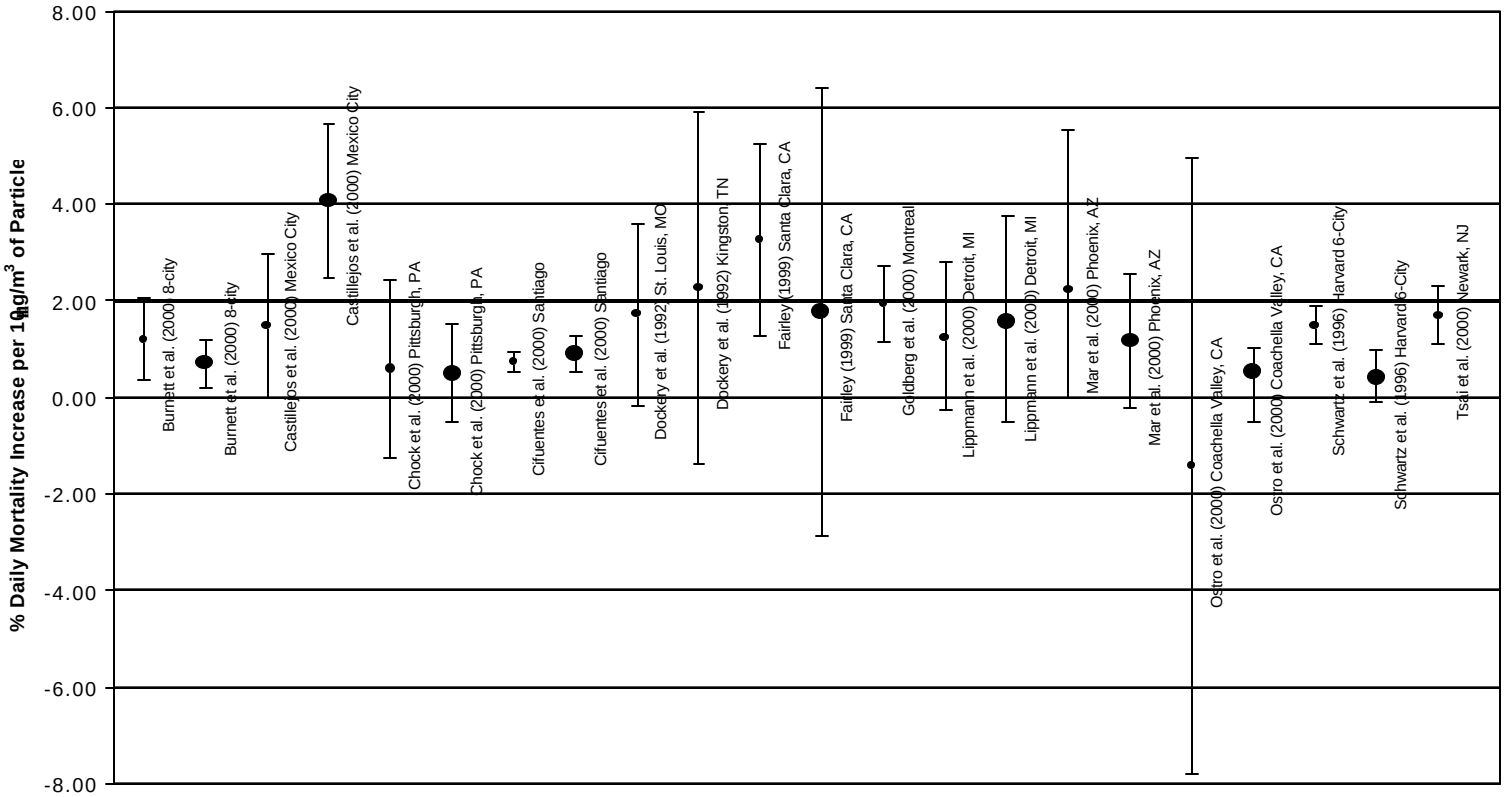
same geographic location using data from 1989 - 1992, there was also an association between PM10 and cardiovascular mortality, although no measures were available for FP or CP (Ostro et al., 1999b).

The more recent Coachella Valley study (Ostro et al., 2000), as well as analyses by Pope et al (1999a) and Schwartz et al. (1999), all indicate, however, that high PM days dominated by windblown dust were not associated with excess mortality. It is not clear whether these findings are due to lower toxicity of crustal particles (relative to those generated by combustion processes) or because people change their behavior and reduce exposure on windy days. Lippmann et al. (2000) examined the effects of different size cuts of PM using mortality data from Detroit and pollution data from the adjacent city of Windsor, Canada. For this study, daily data were collected from May to September with every third- or sixth-day data during the rest of the year, over a two-year period. No associations were reported between all-cause mortality and any PM metrics. However, for cardiovascular mortality, associations were reported for CP, but not FP. Finally, in a study of four years of data from Mexico City, CP had a larger impact and stronger association than FP for all-cause, cardiovascular and respiratory mortality (Castillejos et al., 2000).

Three separate studies of PM-mortality relationships in Phoenix also demonstrate effects from exposure to CP. Mar et al. (2000) found stronger associations of all-cause mortality with CP than with FP for individuals 65 and older. Equally strong associations were reported linking both FP and CP with cardiovascular mortality. Using a different statistical model, Smith et al. (2000) also found stronger associations and estimated effects between all-cause mortality and CP, relative to FP. Similarly, Clyde et al (2000) also reported stronger effects for CP in their analysis of data from Phoenix. Finally, Wichmann et al. (2000) analyzed several years of mortality data from Erfurt, Germany. Most of the analysis was focused on PM data using a mobile aerosol spectrometer, which provided size-specific number and mass concentration data in several size classes. However, filter-based impactor data on PM10 and FP were collected at the same time. The analyses of these data indicated an association between all-cause mortality and PM10, but not with FP.

More mixed results are generated from an analysis of FP and CP data from Santiago, Chile (Cifuentes et al., 2000). The authors report that the results were season-dependent. FP had a stronger association with mortality for the year as a whole and in the winter, whereas CP had a stronger effect during summer. Lipfert et al. (2000) analyzed data from Philadelphia and the surrounding metropolitan area. For all-cause mortality in Philadelphia, the results indicate stronger associations (based on t-statistics) for FP than CP, but the effects per $\mu g/m^3$ are of similar magnitude for the two measures. For cardiovascular mortality in the seven-county region, FP had a stronger association and effect size than CP, while for respiratory mortality, the effect size for CP was greater. Finally, in a relatively small data set from Pittsburgh, Pennsylvania, Chock et al. (2000) report no clear association between mortality and either FP or CP for individuals under 75 years old.

1 Figure 7.4 Daily Mortality Increases Associated with Fine and Coarse Particles



2

3

4 Note: Bar represents 95% confidence interval; small and large dots represent fine and coarse particles respectively.

In summary, the relative results of FP versus CP, as summarized in Table 7.2 and Figure 7.4, are mixed. In some of the mortality studies, primarily those undertaken in cities on the East Coast, FP effects appear to dominate. In other studies, CP has a stronger association with mortality, while in a third set of studies, the effects of FP and CP are approximately similar. However, on average, the effect of a unit increase in FP appears to be greater than that of CP. For the studies summarized in Table 7.2 and Figure 7.4, the average effect of FP is about 1.7% per 10 $\mu g/m^3$, with a range of around 0.6 to 5.2%. For CP, the mean effect of the summarized studies is around 1% per $\mu g/m^3$ with a range from less than zero at the low end to 2 to 4% on the high end. Thus, the FP effect generally appears stronger per unit mass due to either greater intrinsic toxicity, greater indoor infiltration rates, or lower exposure measurement error.

7.3.3 Effects by Chemical-specific or Source-oriented Analysis and by Other Size Cuts

Besides examining the relative impacts of FP and CP, several studies have examined the effects of chemical-specific constituents, including sulfates and a wide range of elements, especially metals. For example, in a study of Santa Clara County, Fairley (1999) examined the impacts of nitrates, sulfates, and COH (coefficient of haze). The latter is highly correlated with elemental carbon, and is likely to be a good marker of pollution from motor vehicles (especially diesel exhaust) and of wood smoke. All three of these constituents of FP were associated with all-cause mortality, while nitrates were also associated with cardiovascular mortality. These findings were consistent with those in the Netherlands where associations were also reported for sulfates, nitrates, and black smoke (Hoek et al, 2000). In a study in Buffalo, Gwynn et al. (2000) reported effects on total mortality for COH, sulfates and hydrogen ion, a measure of aerosol acidity. Lippmann et al. (2000) did not find associations of mortality with sulfate or hydrogen ion in Detroit, although only limited data for these pollutants were available. In their study of the eight largest Canadian cities, Burnett et al. (2000) examined the impact of 47 separate elements within FP and CP. Among the constituents in the fine fraction, sulfates, zinc, nickel and iron were all associated with mortality, as was COH. These elements emanate from a wide range of sources, including, among those relevant to California, oil combustion, road dust, tire wear, and incinerators (Burnett et al., 2000).

Several studies also examined source-oriented combinations of pollutants. For example, Ozkaynak and Thurston (1987), used 1980 U.S. vital statistics data in a cross-sectional analysis of air pollution and mortality. Applying fine particle source apportionment techniques, particles from industrial sources (e.g., iron and steel emissions) and from coal combustion were more significant contributors to mortality than were soil-derived particles. Laden et al., (2000) examined FP data from the Harvard Six-City study, and characterized the pollutants into three different factors: motor vehicle emissions, coal combustion, and soil and crustal material. Generally, both the motor vehicle and coal factors were associated with mortality with the strongest effect from the former. The crustal material in FP was not associated with mortality. In a study with a limited number of days in three New Jersey cities, Tsai et al. (2000) examined the effects of source-type components on mortality. Using factor analysis, this study reported associations of sulfates and motor vehicle tracers with both all-cause and cardiopulmonary mortality. Ozkaynak et al. (1996a) also reported associations between pollutants linked with motor vehicles and total, cardiovascular and respiratory mortality.

Finally, Wichmann et al. (2000) examined the effects of FP mass as well as ultrafine particles (0.01 to 0.1 μm) for the small German city of Erfurt. For ultrafines, the number of particles rather than the mass is used as the exposure measure. For this study, three different size classes of ultrafines were measured, including 0.01 to 0.03 μm, 0.03 to 0.05 μm, and 0.05 to

0.1 μm. The authors found that both FP mass and several measures of ultrafines were associated with daily mortality.

Several studies have indicated a potential role for high concentrations of acidic sulfates in excess human mortality, particularly in London in the 1950s and 1960s (Thurston et al., 1989, Ito et al., 1993). More recent studies of cities in North America, with lower acidic sulfate levels have been inconsistent (Dockery et al., 1992; Burnett et al., 2000; Lippmann et al., 2000, Gwynn et al., 2000). For instance, Dockery et al. (1992) found that PM10 concentrations showed a stronger relationship with daily mortality in St. Louis than did ambient sulfate levels. As noted above, Lippmann et al. (2000) also did not find an association of sulfates with mortality in Detroit. In contrast, Gwynn et al. (2000), in a time-series analysis in Buffalo, NY, found stronger relationships between both acid particles and sulfates and respiratory mortality than that observed for PM10. However, she and her colleagues found no relationship between sulfates and circulatory (cardiovascular) mortality. Burnett et al. (2000) found associations between sulfates and mortality in eight Canadian cities. Thus, strong acid sulfates may play a role in the observed PM-mortality associations, particularly in urban areas with elevated levels of these sulfate classes. However, it should be noted that in California, strong acidic sulfates (particularly sulfuric acid) constitute but a small fraction of PM mass (Chapter VI).

7.3.4 Mortality Displacement

Additional support for pollution-related mortality occurring outside of the hospital and for the likelihood of significant shortening of life is provided by recent studies reporting associations between ambient PM and increased heart rate, decreased heart rate variability, and the incidence of arrhythmias (Liao et al., 1999; Pope et al., 1999b; Peters et al., 2000; Gold et al., 2000; see section 7.7). These outcomes are considered reliable predictors of death from heart disease (Tsugi et al., 1996; Nolan et al., 2000). Direct evidence for a nontrivial reduction in life expectancy is provided by studies that statistically control for the phenomenon of mortality displacement; i.e., in which the time of death might be delayed by only a few days. If all pollution-related deaths were associated with such mortality displacement, the total life shortening would likely be very small. However, both Schwartz (2000c) and Zeger et al. (1999) have shown, using both frequency- and time-domain methods, that most air pollution-associated mortality is not due to such displacement. For cardiovascular deaths, mortality displacement does not appear to be a major factor, as the average life-shortening appears to be greater than two to three months. In contrast, deaths from chronic obstructive pulmonary disease (COPD, which consists mainly of emphysema and chronic bronchitis) appeared to be more consistent with a mortality displacement hypothesis (Schwartz, 2001a, 2000c).

The likelihood of significant loss in life expectancy is reinforced by studies indicating that death occurring outside of a hospital had larger (two- to four-fold) and stronger associations with PM than did deaths occurring inside hospitals (Schwartz 2001a; Schwartz 2000c). This suggests that some of the impacts of PM occur among a subgroup that is not under intensive medical care, and therefore may not necessarily be at the end-stage of their disease. However, it is possible that some out-of-hospital deaths may have occurred among the large contingent of uninsured people in the U.S., who perhaps should have been under medical care.

Finally, evidence of a significant loss in life-years from air pollution is provided by studies of infants and children (see Section 7.7.3). Several recent studies suggest that exposure to PM may result in neonatal or infant mortality (for example, Woodruff et al., 1997; Ostro et al., 1999a; Bobak and Leon, 1998). These studies indicate that infants and children, possibly with

pre-existing respiratory illness, may represent an additional subgroup that is especially sensitive to effects of exposure to ambient PM pollution.

7.3.5 Analysis of Thresholds

For short-term exposure to PM, two general methods are available to address the issue of the existence of a threshold, or an ambient PM level below which there would be no risk of a significant adverse health outcome. First, it can be examined indirectly, by considering data sets with very low mean ambient concentrations. Secondly, it can be examined directly by developing statistical tests that carefully model the shape of the concentration-response function. Both of these approaches appear to indicate the lack of an observable population threshold. Regarding the first method, several studies have been conducted in cities with low ambient concentrations of PM10, including Morgan et al (1998) for Sydney, Australia (mean = 18 $\mu g/m^3$, based on conversion from co-located nephelometry data), Wordley et al. (1997) for Birmingham, UK (mean = 26 $\mu g/m^3$), Schwartz et al., (1996) for the Harvard Six-Cities (mean = 25 $\mu g/m^3$), Burnett et al. (2000) for the eight largest Canadian cities (mean =26 $\mu g/m^3$), and Gwynn et al. (2000) for Buffalo and Rochester (mean = 24 $\mu g/m^3$). In addition, several cities in the data set used by Samet et al. (2000a) have mean concentrations in the low 20s. Examination of these data indicates that the concentration-response functions are not driven by the high concentrations and that the slopes of these functions do not appear to increase significantly at higher concentrations.

Among the statistical approaches, Schwartz et al. (2000a) simply examined the concentration-response relationship in 10 U.S. cities, restricting the data to only days where PM10 < 50 $\mu g/m^3$. The resulting risk estimates were statistically significant and greater than for that of the entire data set. Two other papers first addressed the issue of whether existing statistical techniques could identify a threshold, assuming one existed. Cakmak et al. (1999) simulated data with varying amounts of exposure measurement error, based on actual data from Toronto. They examined whether statistical models used in most air pollution epidemiology (including locally weighted smoothing techniques in Poisson regression models) would be able to detect thresholds in the PM-mortality association. They concluded that, if a threshold existed, it is highly likely that the existing statistical modeling would detect it. Many mortality papers have, in fact, examined the shape of the concentration-response function and indicated that a linear (non-threshold) model fit the data well (Pope, 2000)

A different statistical approach was used by Schwartz and Zanobetti (2000) in their analysis of 10 U.S. cities. The authors combined concentration-response curves across the cities, after demonstrating that this approach produced unbiased estimates. Predicted values of the response function were estimated at 2 $\mu g/m^3$ intervals. Results from this approach did not provide any evidence for a threshold effect. Finally, Daniels et al. (2000) used an alternative statistical approach to test for the existence of a threshold using the 20 largest cities in the U.S. The authors considered three alternative log-linear regression models. One used a simple linear term for PM10, which could then be used as a basis for comparison with the other models. A second model used a cubic spline model that would allow for non-linearity in PM10 that could represent a threshold function. The third model presumed a threshold, in which a grid search would be used to test for a concentration that would support a threshold. The results indicated that for the second model, which can allow for a threshold if the underlying data suggest one, a linear specification provided the best fit to the data. Second, analysis using the grid search model suggested that no threshold was apparent for either total mortality or cardiopulmonary mortality. Finally, using a goodness-of-fit test (Akaike's information criterion) to compare the simple linear no-threshold model with models that would

allow for a threshold concentration, the authors reported that there was no evidence to prefer the threshold models to the linear model.

7.3.6 Summary of Findings

Staff concludes the following from the above results:

- The observed associations between daily changes in PM10 and mortality appear to be independent of the effect of weather factors, seasonality, time, and day of week – all of which are typically controlled for in the analyses. The studies include a wide range of environments, pollution-temperature conditions, population age distributions, background health conditions, socioeconomic status, and health care systems. The range of effect is approximately 0.5% to 1.6% increase in mortality per 10 $\mu g/m^3$ of PM10. However, when longer exposure averaging times are examined, using distributed lags of several days or cumulative exposures of up to several months, the estimated effects may be approximately 2% per 10 $\mu g/m^3$. Although the relative risk per unit is low, the large number of people exposed suggests the existence of a potentially major impact on public health.

- The effects of PM cannot be explained by exposure to other pollutants. As might be expected, examining several co-varying pollutants in the same model typically increases the variation in the estimated PM effect. However, the estimated PM impact is generally consistent regardless of the concentration of, or degree of co-variation with, other pollutants, giving strong support to an independent effect of PM.

- The elderly, those with chronic heart or lung disease, and infants appear to be at significantly greater risk of PM-associated mortality. The results indicate that much mortality associated with acute exposure is not the result of just a few days of life shortening. Rather, for cardiovascular mortality, it appears that significant reductions in life expectancy may be involved. In addition, if the associations between PM and infant mortality represent causal relationships, significant reductions of life expectancy could result.

- The effects associated with short-term exposure to PM appear to occur at current ambient concentrations, including many cities or counties where the mean PM10 concentration is around 25 to 35 $\mu g/m^3$ (Figure 7.1). As suggested by Figure 7.2, greater uncertainty is apparent with decreasing concentrations, particularly those below about 25 $\mu g/m^3$.

- No threshold of response has been observed in the PM-mortality studies. Several direct and indirect approaches have consistently found that non-threshold, linear models provide the best fit to the data.

- Premature mortality appears to be associated not only with PM10, but also with both fine and coarse particles, as well as sulfates (a subset of FP). The effects per unit mass appear to be greater for FP than CP; this may be due to intrinsically greater toxicity of FP versus CP, but may also be attributable to differential measurement error in monitoring for CP than for FP, or greater indoor infiltration rates of FP versus CP (and therefore greater overall exposure to FP), or to some combination of these three. In addition, there is preliminary evidence that pollutants from mobile sources, oil burning, steel industry emissions, and coal combustion are associated with mortality. Crustal materials, particularly those entrained on windy days, are less likely associated with premature mortality.

7.4 Chronic Exposure – Mortality

7.4.1 Study Design and Methods

Several air pollution studies examine the effects of long-term exposure to PM using a prospective cohort design. In this type of study, a sample of individuals are selected and followed over time. For example, Dockery et al. (1993) published results for a 15-year prospective study based on approximately 8,000 individuals in six cities in the eastern United States. Pope et al. (1995) published results of a 7-year prospective study of the mortality experience of approximately 550,000 individuals in 151 cities in the United States using a cohort participating in a long-term investigation sponsored by the American Cancer Society (ACS). These studies used individual-level data so that other factors that affect mortality can be characterized and adjusted for in the analysis. Specifically, these studies were able to control for mortality risks associated with differences in body mass, occupational exposures, smoking (current and past), alcohol use, age, and gender. Once the effects of individual-level factors were determined, the models examined whether longer-term city-wide averages in PM (measured as PM10, PM2.5 or sulfates) were associated with different risks of mortality and life expectancies. Several different cause-specific categories of mortality were examined, including lung cancer, cardiopulmonary, and all other causes. These studies incorporate much, but not all, of the impact associated with short-term exposures (Kunzli et al., 2001). An effect that would tend not to be included in the long-term studies is mortality displacement of a very short-term nature, such as a few days. These effects would not alter the differences in overall life expectancy observed in the longer-term studies.

Statistical analysis used proportional hazards regression modeling (Cox, 1972) with time since enrollment as the underlying time variable. The study samples were stratified by combinations of age (5-year groups), gender and race. Additional analyses were undertaken after stratifying the samples by smoking habit and gender. The greatest uncertainties in these studies involve the disease-relevant times, durations, and intensities of exposure. Both studies assigned city-wide, multi-year averages that occurred when the study participants were young to middle-aged adults (between ages 20 and 50, approximately). Thus, early childhood exposure was not estimated and no within-city differences in exposure were incorporated into the analysis. These errors in exposure assessment would tend to make it more difficult to detect an effect of pollution and would bias the analysis towards the null hypothesis of no effect. Therefore, it is unlikely that bias or misclassification of exposure could explain the results.

7.4.2 Summary of Findings

Both the ACS and Harvard Six-Cities studies report robust and statistically significant associations between several years of exposure to PM and various measures of mortality. Smoking was the dominant factor in explaining mortality patterns, overall and for each of the cause-of-death categories. Regarding air pollution effects, Dockery et al. (1993) reported associations between total mortality and PM10, PM2.5, and sulfates. An association with CP is also apparent (USEPA, 1996). Smaller associations were found with total suspended particles, sulfur dioxide, nitrogen dioxide, and aerosol acidity, but no association was found with ozone. Using a model that included smoking and other non-pollution explanatory variables, all-cause mortality and cardiopulmonary deaths (but not “all other causes”) were both related to sulfates and PM2.5. In additional analyses, PM2.5 was associated with cardiopulmonary mortality but not with “all other” mortality. In this study, PM2.5 concentrations ranged from 11 to 29.6 $\mu g/m^3$ (with a mean of 18 $\mu g/m^3$) and PM10 ranged from 18 to 46.5 $\mu g/m^3$ (with a mean of 30 $\mu g/m^3$). It should be noted that these pollutants were measured only for part of the follow-up time for this cohort: while the mortality experience in the Six Cities

covered the years 1974 – 1991, PM2.5 and PM10 were measured from 1979 through 1985, while sulfates were measured from 1979 through 1984. During these pollutant measurement periods, the concentrations of PM2.5 and sulfates remained relatively stable; nevertheless, the effects of exposures prior to the study could not be evaluated with this data set.

In the study using the ACS cohort, Pope et al. (1995) reported associations between fine particles and sulfates with both all-cause mortality and cardiopulmonary mortality. Across the 50 cities with FP data, FP ranged from 9 to 33.5 $\mu g/m^3$, with a mean of 18.2 $\mu g/m^3$. For the 151 cities with sulfate data, sulfates ranged from 3.5 to 23.5 $\mu g/m^3$ with a mean of 11 $\mu g/m^3$. Exposure data collection was not concurrent with the mortality incidence data: annual arithmetic mean sulfate data were obtained for the year 1980, while for FP the investigators used the city-specific medians of data collected from 1979 to 1983. Mortality among the cohort, meanwhile, was assessed from September 1982 through 1989. The relative risk estimates for this study were smaller than those reported from the Six-Cities study but the confidence intervals around the relative risk estimates overlapped enough that the results were statistically indistinguishable. The estimated mortality effects of approximately 4 to 7% per 10 $\mu g/m^3$ of long-term exposure to PM10 are much larger than those effects associated with daily exposure (approximately 1% per 10 $\mu g/m^3$). These studies also provide a basis for calculating reductions in life expectancy associated with PM exposure. The results suggest that the 24 $\mu g/m^3$ difference in PM2.5 between the cleanest and dirtiest cities is associated with almost 1.5 years difference in life expectancy in the two cities (Pope 2000). Brunekreef (1997) used a life table for men in the Netherlands and estimated a difference of 1.1 years in life expectancy between the two extreme cities in the ACS study. In addition, the difference in life expectancy of a person who actually died from diseases associated with air pollution was estimated to be about 10 years. This is because air pollution-related deaths only make a small fraction of the total deaths in a given city. Subsequent analysis by the authors (reported in Krewski et al., 2000) demonstrated an association between mortality and PM, when PM2.5 was used as the metric of exposure. No association was found, however, when either PM15 or the coarse particle fraction measured as PM15 – PM2.5 was used.

Chronic exposure to PM was also examined using a smaller and younger non-smoking cohort participating in the Seventh Day Adventist Health Study (Abbey et al., 1999). For the years prior to 1987, PM10 data were unavailable and were estimated from TSP concentrations. In this study, neither mean PM10 nor sulfate concentrations were associated with mortality. However, an interquartile range difference of 43 days when PM10 levels were greater than 100 $\mu g/m^3$ was associated with both all-cause and nonmalignant respiratory mortality in males, but not females. In a follow-up study using a subset of the cohort living near airports, estimates of PM2.5 were developed from data on airport visibility (McDonnell et al., 2000). PM10 was again estimated from season- and city-specific regressions using TSP data. Positive but non-statistically significant associations were found between all three measures of PM (PM10, CP and FP) and both all-cause and respiratory mortality in males. Although the mean of the estimated value of PM10 was relatively high in these studies (i.e., 51 $\mu g/m^3$ in Abbey et al, 1999 and 59 $\mu g/m^3$ in McDonnell et al, 2000), most of the measures of PM10 were estimated from either TSP or from airport visibility. This process added errors in the measurement of exposure which would likely lead to a lowered effect estimate.

Krewski et al. (2000) completed an independent validation and reanalysis of both the Six-Cities and the ACS cohort studies. The first task of this study was to recreate the data sets and validate the original results. Krewski et al. (2000) reported few errors in the coding and data merging in the original studies and basically replicated the results of both studies. The second task was to conduct an exhaustive sensitivity analysis of the original studies to determine whether the results were robust. Specifically, the authors examined the effects of:

(1) alternative statistical models; (2) potential individual-level interactions and confounders such as physical activity, education, body mass, smoking status, marital status, alcohol consumption and occupational exposure; (3) potential city-wide confounders such as population growth, income, weather, number of hospital beds and water hardness; (4) consideration of various subgroups; (5) non-linear specifications in the dose-response function that would allow for the possibility of a threshold; (5) co-pollutants including ozone, sulfur dioxide and nitrogen dioxide; (6) alternative PM exposure estimates, including different years and particle sizes, (7) underlying variation from city to city; (8) spatial correlation between cities; and (9) time-dependent variables such as air pollution exposure and individual risk factors that change over time. In general, the re-analysis confirmed the original results of associations between mortality and long-term exposure to PM.

Among the more important new findings were: (1) education (possibly serving as a marker for socioeconomic status, health care or lifestyle factors associated with SES) appears to be a significant effect modifier (see Section 7.7.2 below); (2) FP was more strongly associated with mortality than was either PM10 or CP; (3) the results were not confounded by either individual-level or city-wide (ecological) covariates; (4) the associations between sulfate and FP and both all-cause and cardiopulmonary mortality were near linear within the relevant ranges, with no apparent threshold; (5) the PM effects were not confounded by and were independent of effects of other pollutants, (6) the effects were robust with respect to alternative functional forms, alternative air pollution data, and detailed spatial analysis; and (7) the results of the original investigators were confirmed.

7.5 Daily Exposure – Morbidity

Over the last decade, several hundred epidemiologic studies have reported associations between alternative measures of PM and a range of morbidity outcomes. The PM measures have included PM10, black smoke (BS), COH, sulfates, and more recently as monitoring data has become available, FP and CP. The health outcomes associated with PM include, but are not limited to, hospitalization for cardiovascular or respiratory disease, emergency room and urgent care visits, asthma exacerbation, acute and chronic bronchitis, restrictions in activity, work loss, school absenteeism, respiratory symptoms, and decrements in lung function. Typically, these studies have involved either of two analytic methods. First, many of the outcomes use a methodology similar to that described above for mortality related to short-term exposure -- time-series analysis of daily count data. Specifically, daily counts of an endpoint such as hospitalization for cardiovascular disease are examined in response to single- and multi-day concentrations of PM. As in the case of mortality, these models also control for potential confounders such as season, meteorology, day of week, and time trends. A second approach involves the use of panel data, where a cohort of subjects (e.g., asthmatic children) are followed prospectively over a period of several months or years while daily health outcomes and pollution measures are recorded and then compared. In the following subsections, we briefly review some of the important health outcomes, with particular attention given to studies undertaken in California. The review is not meant to be exhaustive but should serve to illustrate the range and consistency of morbidity effects associated with PM10 or its components.

7.5.1 Cardiovascular Hospital Admissions

Associations between daily concentrations of PM10 and daily hospital admissions for cardiovascular disease have been reported for close to a hundred cities in the U.S, Canada and Europe (Table 7.3). As is the case for the mortality studies related to short-term exposure, there are several multi-city efforts (Schwartz et al., 1999; Samet et al., 2000a; Zanobetti et al., 2000a). For example, Schwartz et al., (1999) examined daily hospital

admissions for cardiovascular disease (ICD9 codes 390 – 429) from 1988 to 1990 among persons above age 65 for eight metropolitan areas, including Chicago, Colorado Springs, Minneapolis, New Haven, St. Paul, Seattle, Spokane, and Tacoma. For five of the cities and for the effect estimate pooled across all eight cities, a statistically significant association was reported with PM10. Across the cities, a 10 $\mu g/m^3$ change in PM10 was associated with about a 1% change in hospitalization for cardiovascular disease. The median PM10 concentration in these cities ranged from 23 to 37 $\mu g/m^3$.

Samet et al. (2000a) examined data on hospitalizations for cardiovascular disease among people 65 and older from 14 U.S. cities from 1985 to 1994. The cities were located throughout the U.S., though none was in California. Again, a statistically significant association was reported for the pooled estimate of the cities with an effect estimate of 1.1% per 10 $\mu g/m^3$. The estimate increased to 1.5% per 10 $\mu g/m^3$ when a two-day average of PM10 was used and PM10 was restricted to concentrations less than 50 $\mu g/m^3$. For these cities, the mean PM10 ranged from 24 to 45 $\mu g/m^3$, with a group mean of 33 $\mu g/m^3$. Zanobetti et al. (2000a) essentially confirmed the Samet et al. (2000a) results and also demonstrated that other pollutants such as carbon monoxide, ozone and sulfur dioxide were not confounding or modifying the estimated effects of PM10. Burnett et al. (1997a) also reported an association between PM, measured as COH, and congestive heart failure (ICD9 = 427) for those ages 65 and above living in Canada's 10 largest cities, from 1981 to 1994. The effect size was similar to that reported for PM10 in the U.S. studies. Similar results have been reported between PM and either total cardiovascular disease or its components (e.g., heart failure or ischemic heart disease) in a disparate range of cities including, but not limited to: Detroit (Lippman et al., 2000), Tucson (Schwartz, 1997), Toronto (Burnett et al., 1997b), London (Atkinson et al., 1999), Edinburgh (Prescott et al., 1998), Sydney (Morgan et al., 1998), Chicago (Morris and Naumova, 1998) and Hong Kong (Wong et al., 1999). In addition, Stieb et al., (2000) reported associations between emergency department visits for angina or myocardial infarction and both PM10 and PM2.5.

Among California cities, associations have been reported between PM10 and hospitalization for total cardiovascular disease, myocardial infarction, congestive heart failure and cardiac arrhythmia among individuals above age 30 in Los Angeles (Linn et al., 2000). Daily gravimetric measures of PM10 were estimated from TEOMs and averaged 37 $\mu g/m^3$ in the winters to 54 $\mu g/m^3$ during autumn. In another study of Los Angeles' hospitals, Moolgavkar (2000b) reported associations between PM10 and total cardiovascular admissions among people ages 20 to 64, and 65 and above. Mean PM10 was 44 $\mu g/m^3$. The effect magnitudes of PM10 estimated for Los Angeles were generally similar to those reported for other studies in the U.S. -- a 0.6 to 2% increase in cardiovascular hospitalizations per 10 $\mu g/m^3$ of PM10.

Only a few cardiovascular admissions studies have measured FP and CP concentrations. However, among those that measured different particle sizes, Lippmann et al. (2000) reported associations of hospitalizations for heart failure and ischemic heart disease with both FP and CP in Detroit. Likewise, Burnett et al. (1997b) found associations between hospitalizations for total cardiovascular conditions, heart failure, dysrhythmias, and ischemic heart disease and both FP and CP in Toronto. Finally, in the Moolgavkar (2000b) study in Los Angeles, an association was reported between FP and cardiovascular hospital admissions for the 20 to 64 age group, and 65 and above. Estimates for CP were not provided. Gwynn et al. (2000) found little evidence of a relationship between PM10 or sulfates and circulatory (cardiovascular) hospital admissions.

In summary, studies over the past several years consistently report associations between PM10 and hospitalization for total cardiovascular disease and several of its specific

components, such as congestive heart failure and ischemic heart disease. These effects have been mostly reported among people above age 65, a group that dominates the prevalence of cardiovascular diseases. For many of these studies, the mean PM10 ranges from 25 $\mu g/m^3$ to 40 $\mu g/m^3$, although studies of cities with reported means below and above this range exist, as well. Most of the studies carefully control for the potential confounding of weather, season time, and co-pollutants. Overall, PM10 is consistently associated with these clinically significant cardiovascular endpoints, with a general effect estimate of between 0.6 to 2% per 10 $\mu g/m^3$. These relatively low risk estimates, however, are shared over a large segment of the population who are constantly exposed to PM and who have pre-existing cardiovascular disease. Based on the few studies that have measured both fine and coarse particles, associations are apparent between hospital admissions for cardiovascular diseases and both of these exposure measures. In these studies, mean FP ranged from 17 $\mu g/m^3$ to 22 $\mu g/m^3$. Finally, as indicated in Section 7.3., associations between daily or multi-day exposure to PM10 and cardiovascular-related mortality have also been reported. In addition, Section 7.8 includes a summary and discussion of several of the other cardiovascular outcomes associated with PM such as changes in heart rate, heart rate variability, arrhythmia, heart attacks, and blood viscosity. The coherence of the mortality and morbidity results provides compelling evidence of an effect of PM.

7.5.2 Respiratory Hospital Admissions

Many studies have also used time-series analysis to report associations between daily PM and hospitalization for respiratory diseases (Table 7.3). Such endpoints have included total respiratory admissions (ICD9 = 460-519) for all age groups for those greater than age 65, and admissions for chronic obstructive pulmonary disease (COPD), pneumonia and asthma. For example, the recent NMMAPS multi-city study (Samet et al, 2000a) examined the association between PM10 and several specific respiratory diseases among a group of individuals age 65 and above. Associations were reported between PM10 and both COPD and pneumonia. Among the 14 cities in the analysis, the mean PM10 ranged from 24 to 45 $\mu g/m^3$. The risk estimates were in the range of 1.5 to 3% per 10 $\mu g/m^3$ of PM10.

Similar findings of an association of PM10 and hospital admissions for total respiratory diseases or its components such as COPD, asthma or pneumonia have been reported for many other cities throughout the U.S. including, but not limited to Minneapolis (Schwartz, 1994c; Moolgavkar et al., 1997), Tacoma (Schwartz, 1995), Cleveland (Schwartz, 1996), Buffalo (Gwynn et al., 2000), Chicago (Zanobetti et al, 2000a), Detroit (Lippmann et al, 2000) and Seattle (Sheppard et al., 1999).

Three separate studies have reported similar associations using data from Los Angeles (Linn et al., 2000; Moolgavkar, 2000b; Nauenberg and Basu, 1999). The Linn et al. (2000) study used pulmonary hospital admissions data from 1992 to 1995 and found positive associations with PM10 (mean = 45 $\mu g/m^3$) for the full year, but especially in the winter. Moolgavkar (2000c) used data on COPD for 1987 through 1995, and reported associations with PM10 (median = 44$\mu g/m^3$), and both FP (median = 22 $\mu g/m^3$) and CP for three different age groups: 0 to 19, 20 to 64, and 65 and above. Finally, Nauenberg and Basu (1999) used data on hospital admissions for asthma from 1991 through 1994. Associations were reported with PM10 (mean = 45 $\mu g/m^3$) in the “wet season” (Jan 1 to March 1) but not the “dry season”. The wet season effect was also stronger among MediCal claimants, suggesting an effect modification by income. Gwynn and Thurston (2001) also reported stronger effects from PM10 and other pollutants on respiratory hospital admissions among those without insurance or on Medicaid versus those with private insurance or Medicare.

Besides Moolgavkar (2000b), a few other studies have reported findings using FP and CP. For example, Lippman et al. (2000) found an association between pneumonia admissions for those age 65 and both FP and CP in Detroit. For COPD, an association was also reported for FP, and less so for CP. Likewise, Burnett et al. (1997b) found associations between hospital admissions for respiratory diseases and both FP and CP in Toronto. The Sheppard et al. (1999) study reported above also found associations between both FP and CP and asthma hospital admissions. Finally, Moolgavkar et al. (2000) found an association between FP and hospital admissions for COPD in King County (Seattle). No results were reported for CP.

Associations have also been reported between PM10 and emergency department and urgent care visits, which may or may not result in hospital admissions. Regarding emergency department visits, for example, in a study conducted in Santa Clara County, California, Lipsett et al. (1997) reported associations between PM10 (mean = 61 $\mu g/m^3$) and emergency room visits for asthma during the winters, particularly on the colder days. Using limited data (only one year), Delfino et al. (1997a) found associations between respiratory emergency department visits and PM10, FP, sulfates and hydrogen ion in Montreal. Norris et al. (2000) analyzed emergency room visits for asthma in Spokane and Seattle. In Spokane two years of data on patients of all ages were used, while in Seattle 16 months of data for asthma cases below the age of 18 were used. Besides PM10, a stagnation index was created, which reflected days with relatively low windspeed. Factor analysis indicated that these days were likely to involve higher concentrations of products of incomplete combustion (including fine particulate elemental carbon) and sulfates. In Spokane, associations were found between emergency asthma visits and the stagnation index, but not with PM10. However, for Seattle both of these metrics were associated with emergency room visits for children.

Several other studies have reported effects for children. For example, Tolbert et al. (2000) examined the effects of air pollution on roughly 6,000 pediatric emergency room visits for asthma during the summers of 1993-1995 in Atlanta. Several different statistical models were used to explore the sensitivity of the results to the model selection. PM10 concentrations (mean = 39 $\mu g/m^3$) were highly correlated with 1-hour maximum ozone (r = 0.75). Associations between daily visits and PM10 were reported, with consistent results across all of the models. Medina et al. (1997) analyzed doctors' house calls for asthma in Paris, France for the years 1991 to 1995. Black smoke (BS) was used as a measure of particulate matter. House calls for asthma were divided into three age groups: all ages, 0 to 14 years, and 15 to 64 years. Ultimately, associations were reported for the full age group of 0 to 64 years, but especially for children below age 14. The effect estimate for children, based on a 4-day moving average of BS was 8 times high than that of the older population. Hajat et al. (1999) reported a similar association in London, England between PM10 and doctor visits for asthma for children below age 14. While the effect size was not as high as in the Medina et al. (1997) study, the strongest effect was found from a multiday average of exposure to PM10. In examining allergic rhinitis, Hajat et al. (2001) reported stronger associations for adults than for children. The associations were also stronger for multi-day averages of PM10. Ostro et al. (1999c) analyzed the association between daily visits to primary health care clinics in Santiago, Chile among children under age 2, and ages 2 to 14. This area is characterized by very high levels of ambient PM10, especially during the winter when inversions are common. For this study, several public clinics around the city were organized to undertake a specific study of urgent care visits for lower and upper respiratory symptoms. An association was found between PM10 and visits for lower respiratory symptoms for both age groups.

Several studies suggest relationships between strong acid sulfates and respiratory hospital admissions. In a time-series study in Buffalo, NY, Gwynn et al. (2000) reported stronger associations between both H+ aerosol and sulfates and respiratory hospital admissions than

those observed for either PM10 or COH. Burnett et al. (1994), in an analysis of urgent daily admissions at 168 acute care hospitals in Ontario, Canada, found significant associations of sulfates (lagged 0 to 3 days) with several respiratory diseases, but not with nonrespiratory conditions. These associations were not significant during the winter, when the sulfate levels tend to be lower. However, during the summer months, sulfates are strongly correlated with both PM2.5 and with H+ ($r>0.8$), so it is difficult to ascribe a "causal" role to any one of these PM constituents.

In summary, studies over the past several years consistently report associations between PM10 and several different measures of hospitalization or urgent care for respiratory diseases. The outcomes include hospitalization for total respiratory disease and several of its components including COPD, asthma and pneumonia. In addition, associations have been reported between PM10 and the need for urgent care including emergency department visits, doctor visits, and public clinic visits. These effects have been reported primarily among elderly individuals, but effects have been also reported among all age groups including children under age 18, and children under age 2. For many of these studies, the mean PM10 ranges from 25 $\mu g/m^3$ to 40 $\mu g/m^3$, although studies of cities with reported means below and above this range exist, as well. Most of the studies carefully control for the potential confounding of weather, season time, and co-pollutants. Overall, PM10 consistently is associated with these clinically significant respiratory endpoints, with a general effect estimate of between 1.25 and 5% per 10 $\mu g/m^3$. Based on the few studies that have measured both fine and coarse particles, associations are apparent between hospital admissions for respiratory diseases and both of these exposure measures. In these studies, mean FP ranged from 17 $\mu g/m^3$ to 22 $\mu g/m^3$. Finally, as indicated in Section 7.3 associations between daily or multi-day exposure to PM10 and respiratory-related mortality have also been reported.

7.5.3 Asthma Exacerbation

Asthma affects more than 15 million Americans, including almost 5 million children, making it the most common childhood illness in the U.S. Asthma prevalence increased 75% from 1980 to 1994 in the United States (Mannino et al., 1998). In a recent analysis of data from the National Health Interview Survey, the prevalence of asthma among children aged 5 – 14 was about 67% higher than for adults aged 35 and above (74.4/1000 vs 44.6/1000, respectively; Mannino et al., 1998). Asthma surveillance data developed by the U.S. Centers for Disease Control and Prevention (CDC) and recent reports on asthma hospitalization by the California Department of Health Services (CDHS, 2000), and King County, Washington, indicate that children, especially young children, may experience severe exacerbations at a greater rate than older children or adults (Mannino et al., 1998; CDHS 2000; Solet et al. 2000). Hospitalization rates for children 0 to 4 years are greater than for all others (49.7/10,000/year for ages 0 – 4 versus a range of 18.0 to 25.5/10,000/year for all other age groups) and is four-fold higher among black children versus white children (CDHS, 2000). Younger and poorer communities tend to have the highest rates of asthma hospitalization (Solet et al., 2000, Claudio et al., 1999). While hospitalization rate data are influenced by a number of factors, including access to health care, these data support the notion that asthma may affect young children more than adults.

In the last few years, many studies have been published on the effects of PM exposure on symptoms and lung functions changes in asthmatics (Table 7.4). These studies typically follow a panel of subjects who record daily health outcomes over several months. A range of outcomes have been measured including specific symptoms (e.g., cough, shortness of breath, wheeze, chest tightness, phlegm), medication use, and lung function changes [e.g., peak expiratory flow rate (PEF), forced expiratory volume (FEV), and forced vital capacity (FVC)]. Concurrent air pollution is recorded along with potential confounders that also change

on a daily basis and might be associated with the health outcome such as weather factors, environmental tobacco smoke (ETS) or wood smoke exposure, activity patterns, time spent outdoors, use of air conditioning, and day of week. Generally, the study of air pollution and asthma is analytically challenging since the disease and its triggers are complex and multidimensional. Several of the studies combine individuals with different levels of asthma severity and medication use, or combine asthmatics and non-asthmatics. Nevertheless, evidence for a fairly consistent (but not universal) effect of PM has emerged over the last several years, including several studies conducted in California.

For example, Ostro et al. (2001) examined the effect of PM10 and PM2.5 on 138 African-American children with current, physician-diagnosed asthma living in Los Angeles from August through October, 1993. Daily reporting of cough, shortness of breath and wheeze, and asthma episodes (i.e., the start of several consecutive days with symptoms) were associated with PM10 (24-hour mean = 52 $\mu g/m^3$) and PM2.5 (12-hour mean = 41 $\mu g/m^3$), but not with ozone. The PM10 effects were slightly stronger than those from PM2.5, with a 10 $\mu g/m^3$ change in PM10 associated with an approximate change in onset of symptom rates of from 5 to 15%. In addition, an association was reported between PM10 and the use of extra asthma medication. These findings supported an earlier study of 83 African-American children with asthma in Los Angeles that indicated an association between PM10 and shortness of breath (Ostro et al., 1995).

Delfino et al. (1997b, 1998) also examined panels of asthmatics living in California. In a summer study, 22 asthmatics, ages 9 to 46, from the semi-rural town of Alpine were followed (Delfino et al., 1997b). Symptoms were not related to PM10 (24-hour mean = 26 $\mu g/m^3$) or any of the other pollutants or bioaerosols measured. However, there was an association between PM10 and inhaler use. Delfino et al. (1998) followed a panel of 24 asthmatics, ages 9 to 17 from August to October, 1995 in Alpine. “Bothersome” asthma symptoms (either cough, wheeze, sputum production, shortness of breath, or chest tightness) were associated with both PM10 (24-hour mean = 31 $\mu g/m^3$) and ozone, with a greater relative effect from PM10. The largest effects of PM10 were on those children not currently on anti-inflammatory medication.

In studies outside of California, Yu et al. (2000) followed 133 asthmatics, ages 5 to 13, living in Seattle. A strong association was reported between asthma symptoms and nephelometry data, which is approximately equal to PM1.0 or particles below one micron in diameter. Vedel et al. (1998) examined 75 physician-diagnosed asthmatic children, ages 6 to 13, living in Port Alberni, British Columbia. Several other groups of non-asthmatics were analyzed as well. For the entire group (n = 206), PM10 (median = 22 $\mu g/m^3$) was associated with increases in both cough and phlegm (7% increase in each per 10 $\mu g/m^3$ PM10), and decreased PEF. Stratified analysis indicated effects only among asthmatic children. No consistent effects were found in the other groups of children. Thurston et al (1997) examined children with asthma at a summer camp in Connecticut. Associations were reported between both sulfates and ozone (which were highly correlated) and asthma symptoms, PEF and bronchodilator use. Data on PM10 were not available. Pope and Dockery (1992) studied two different cohorts of fifth- and sixth-grade students in Utah Valley. One group had symptoms of asthma or had been diagnosed by a physician as having asthma, but were not currently on medication. The other group had no history or symptoms of asthma. Associations were found for both groups between PM10 and both PEF and respiratory symptoms. The symptomatic group demonstrated a greater effect from exposure to PM10.

Several studies on asthma have also been completed outside of the U.S. and Canada. For example, Gielen et al. (1997) reported associations between PM10 and both asthma

symptoms and PEF among children in Amsterdam. Hilterman et al. (1998) reported associations between PM10 and symptoms but not PEF in asthmatic adults living in Leiden, Netherlands, while Peters et al. (1997) reported associations between various measures of PM and both symptoms and PEF among adults in Erfurt, Germany. Finally, Romieu et al. (1996) also reported associations between PM10 and asthma exacerbation among a panel of children living in Mexico City.

Overall, the effects of PM on asthma exacerbation are not as consistent as those found with hospitalizations for cardiovascular or respiratory disease. This is likely due to the complexity and multi-dimensional aspects of the disease itself, and the subsequent difficulties in estimating the impact of air pollution. Nevertheless, several well-conducted prospective cohort studies, often involving over 100 children with asthma, have found associations between PM10 and a range of asthma symptoms or medication use. Most of the studies have controlled for potentially confounding factors such as weather and other pollutants, such as ozone. Given the findings reported above, of an association between PM10 and hospitalizations and urgent care for asthma, it is reasonable to expect an impact on less severe asthma outcomes as well.

7.5.4 Respiratory Symptoms and Other Adverse Outcomes

Panel studies and other analytical study designs have also been used to examine the effect of air pollution on the general population (including both asthmatic and non-asthmatic individuals) (summarized in Table 7.4). A wide range of outcomes has been studied including upper and lower respiratory symptoms (in aggregate form and separated out by specific symptoms), lung function changes, restrictions in activity due to respiratory illness, school absenteeism and work loss. Although these effects are clearly not as significant as mortality and hospitalization, they may have an important effect on public health since they impact a greater proportion of the population. Some of these studies are summarized below to provide a sense of the range of impacts associated with exposure to PM.

In a study in three cities in Southern California (Azusa, Glendora and Covina), Ostro et al (1993) examined the daily effects of air pollution on 321 nonsmoking adults. Associations were reported between both sulfates and ozone on lower respiratory symptoms. Schwartz and Neas (2000) reanalyzed three different panel studies to examine the relative impact of FP and CP on respiratory symptoms and peak flow in young children. First, daily respiratory data from 1,844 children in second through fifth grade from six eastern cities (the Harvard Six-Cities) were used. The second and third data sets involved daily data collected from June through August from fourth and fifth grade children living in Uniontown and State College, PA. In both of these studies, twice daily PEF measures were recorded. The analysis of the Six City data suggested that, using single pollutant models, lower respiratory symptoms (any day with at least two of the following: cough, phlegm, chest pain or wheeze) were associated with both FP and CP, as well as sulfates. The stronger effects were observed for FP and sulfates. When considering only cough as the outcome, associations were again found with all of the measures of PM, but the strongest effect was with CP. In the analysis of PEF in the two other cities, an association was found with FP and sulfates but not with CP. Zhang et al. (2000) examined respiratory symptoms among 673 mothers living in Vinton, VA during the summer of 1995. Of all the pollutants considered, only CP were associated with a new episode of rhinitis.

Tiitanen et al. (1999) examined the association between PM and PEF and cough among 49 children with chronic respiratory symptoms living in Kuopio, Finland. Several different measures of PM were available, including PM10, FP, CP, black carbon, resuspended road dust, and ultrafines. Associations were reported between morning PEF and all of the

measures of PM. In addition, incidence of cough was also associated with all of the PM measures. For cough, however, the strongest association was with a 4-day cumulative average of both FP and ultrafines. Since the PM measures were highly correlated, it is difficult to attribute the effect to any single constituent. Schwartz et al. (1994) examined the respiratory symptoms of 300 elementary school children from April to August in each of six eastern cities. Several different endpoints were considered, including lower respiratory symptoms (reports of at least two among cough, chest pain, phlegm, and wheeze), upper respiratory symptoms (reports of at least two among hoarseness, sore throat, and fever), and cough alone. An association was reported between both PM10 and PM2.5 and lower respiratory symptoms, cough, and to a lesser extent, upper respiratory symptoms.

Two studies in the Netherlands examined the impact of wintertime PM10 on symptoms in two panels of children. Boezen et al. (1999) studied a panel of children ages 7 to 11 to determine if those with bronchial hyperresponsiveness (BHR) and high serum concentrations of IgE were more responsive to air pollution. Based on data from three winters, an association was found between PM10 and lower respiratory symptoms among children with BHR and high total IgE. No associations were found among children who did not have both of these factors. The wintertime PM10 averages for the three years were 55, 42 and 31 $\mu g/m^3$. In a related study, van der Zee et al. (1999) examined PEF and respiratory symptoms among children in urban and rural areas with and without asthma, chronic cough, or wheeze (classified as symptomatic). In both the urban and rural areas, associations were found between PM10 and both lower respiratory symptoms and decrements in PEF among the symptomatic children. However, stronger effects were observed in the urban areas. Among the non-symptomatic children, no association between PM10 and symptoms was found. In the urban area, PM10 averaged 48, 37 and 29 $\mu g/m^3$ during the three winters that were studied, versus 35, 35 and 24 $\mu g/m^3$ in the rural area.

Regarding changes in lung function, Hoek et al (1998) reanalyzed data on PEF from four other studies conducted in Utah, the Netherlands, and Uniontown and State College, PA. This paper focused on explaining significant decrements in PEF, defined as a daily change greater than 10% below a person's mean. This change was found to be associated with changes in PM10.

Besides respiratory symptoms and changes in lung function, other less severe symptoms have been reported for the general population. For example, Ostro (1987) and Ostro and Rothschild (1989) used data from six years of the annual Health Interview Survey conducted by the National Center for Health Statistics. Based on a two-week recall period, the endpoint used in these studies was restricted activity days, which includes days spent in bed, days missed from work, or days when activities were partially restricted due to illness. In Ostro (1987), which included 49 metropolitan areas, an association was reported between FP, estimated from airport visibility and restricted activity in adults. Ostro and Rothschild (1989) reported an association between FP and both respiratory-related restrictions in activity and minor restrictions (days where activity was restricted but not resulting in work loss) in adults. These studies imply about a 10 to 15% change in reduced activity per 10 $\mu g/m^3$ of PM10. Finally, Ranson and Pope (1993) examined PM10 and weekly absenteeism in an elementary school in Utah. An association was reported with PM10, with about a 4% increase in absenteeism per 10 $\mu g/m^3$.

7.6 Chronic Exposure – Morbidity

Data from the past quarter century suggest that long-term PM exposures are associated with chronic respiratory symptoms or disease, and possibly with decreased lung function. Much of this evidence derives from cross-sectional analyses, which compare symptom or disease

prevalence, or lung function, during a given time period (e.g., one year) among communities with different average pollution levels (e.g., Ferris et al., 1973; 1976; Hodgkin et al., 1984; Mullahy and Portney, 1990). Cross-sectional studies, however, while suggestive of potentially meaningful associations, are generally not considered good evidence of causal relationships because inter-city differences may be due to unmeasured factors other than air pollution. Also, chronic health effects are thought to occur as a result of long-term or repeated exposures, but cross-sectional investigations generally present a snapshot in time and are not informative regarding the critical exposure averaging time (e.g., 1 year, 10 years, or even the number of times a given level is exceeded during a specified period). Nevertheless, several large cross-sectional investigations in the U.S. and Europe, in which individual-level data on a variety of other relevant factors have been collected (e.g., smoking status, exposure to environmental tobacco smoke, occupational exposures), provide reasonably consistent evidence for effects of long-term exposure to PM on chronic respiratory health outcomes.

Several large-scale cohort studies provide prospective evidence related to long-term effects of PM exposure. These studies have collected information on individual participants, and therefore can statistically control for most of the potentially relevant confounding variables, including cigarette smoking, exposure to environmental tobacco smoke, occupational exposures (for adults), weight, alcohol consumption, and so forth. The most important of the relevant cross-sectional and cohort studies are summarized in the following paragraphs. Most have been conducted in the United States, and several have been undertaken (at least in part) in California. One large cohort study undertaken in four cities in the Los Angeles basin (the Chronic Obstructive Respiratory Disease or CORD study) is not included in the discussion, because inter-city differences in participants' lung function were not presented by pollutant (e.g., Tashkin et al., 1994; Detels et al., 1991).

7.6.1 The Adventist Health Study

In 1977, a cohort of 6,338 nonsmoking non-Hispanic white Seventh Day Adventists, aged 25 years and older and residing principally in three large metropolitan areas of California (San Francisco, San Diego, and the South Coast Air Basin), were enrolled in a long-term study of the effects of air pollution on respiratory health (AHSMOG). Approximately 10% of the study population lived in other areas of California. One criterion for enrollment was residential stability: all participants had to have lived within 5 miles of their 1977 address for 10 years or longer. Participants completed questionnaires in 1977, 1987, and 1992 regarding residential and work location histories, past smoking, exposure to ETS, occupational exposures, presence of various respiratory symptoms, and physician diagnoses of respiratory disease. Cumulative air pollution exposure was assessed by interpolation of fixed-site monitoring data in relation to the subjects' residences and worksites during the study period. Numerous reports describing the morbidity and mortality of this cohort have been published: earlier reports focused on total suspended particulates (TSP) as the PM metric (e.g., Abbey et al., 1993) and will therefore not be discussed. Several of the more recent articles are described below, while the mortality results are described in Section 7.3.

Abbey and colleagues (1995a,b) analyzed the incidence of chronic respiratory disease in relation to several particle metrics for the 10-year period 1977 through 1987 for a subset of 3,914 study participants. PM10 concentrations were estimated using site- and season-specific regressions on TSP data during this period. They reported that long-term exposures to estimated PM10 concentrations exceeding 80 or 100 $\mu g/m^3$ for at least 250 hours/year produced statistically significant increases in risk of newly reported symptoms of overall airway obstructive disease (AOD, consisting of a triad of asthma, chronic bronchitis and emphysema) and of chronic bronchitis alone, but not asthma (Abbey et al., 1995a). Although point estimates of risk associated with lower concentrations of estimated PM10 were all

greater than one, none was statistically significant. For a subset of the cohort living near airports (n=1,868), PM2.5 concentrations were also estimated using visibility data (Abbey et al., 1995b). In this group, PM2.5, PM10 and sulfates were all significantly related to worsening severity of AOD (relative risks of 2.20, 2.64, and 3.04, respectively) or asthma alone (relative risks of 2.05, 2.82, and 2.75, respectively), while sulfates and PM10, but not fine particles, were both associated with significantly increased risks of AOD, and PM10 with chronic bronchitis. All of the long-term studies in this document involve exposure measurement error, which generally would tend to impede researchers' ability to detect any relationship between air pollution and health. In these reports this situation is exaggerated because neither PM2.5 nor PM10 were directly measured, suggesting that these results, though perhaps reliable qualitatively, should not be considered quantitatively accurate.

Beeson et al. (1998) examined associations between several air pollutants and lung cancer incidence (n=36 incident cases, 16 in men and 20 in women) from 1977-1992), adjusting for several covariates (attained age, pack-years of past cigarette smoking, years of education, and consumption of alcohol at baseline), though a variety of other variables were also examined as potential confounders. The estimated annual mean concentration of PM10 from 1973-1992 was 51 $\mu g/m^3$ (SD=16.52). As in prior reports on this cohort, PM10 concentrations from 1977-87 were estimated from TSP measurements, while after 1987 PM10 was measured directly. Incident lung cancer in men was significantly associated with the average annual mean concentration of PM10 (RR = 5.21, 95% C.I.=1.94-13.99, for an interquartile range or IQR of 24 $\mu g/m^3$), with somewhat lower estimates for ozone and SO_2. For women, lung cancer incidence was associated with PM10 (including the annual mean concentration and several exceedance frequencies), but these relationships were not statistically significant. In multi-pollutant models, the coefficients for PM10 and SO_2, but not ozone, remained stable. Although these RR estimates for men were stable in a variety of sensitivity analyses, they are substantially higher than those observed in other investigations, and may be due to a lower baseline lung cancer rate in the nonsmoking Seventh Day Adventist source population. However, the relatively small number of cases on which these are based suggests a need for cautious interpretation.

In 1993, 1,391 of the study participants who had completed all three questionnaires and met several other criteria successfully completed lung function testing. For this analysis, mean PM10 levels averaged over monthly values from 1973-1993 were 54.1 $\mu g/m^3$ for male subjects (range 20.0 – 80.6) and 52.7 $\mu g/m^3$ (range 21.3 – 80.6) for female subjects. An inter-quartile difference of 54 days/yr in excess of 100 $\mu g/m^3$ PM10 was associated with significant decreases in FEV_1 of –7.2% (95% C.I = -11.5 - -2.7) in men whose parents had a history of obstructive lung disease or hay fever, and of –1.5% (95% C.I. –2.7 - -0.4) FEV1/FVC in male never-smokers. No such effects were seen in women or in other strata of men. These results should be viewed with caution because: (1) these results are essentially cross-sectional and represent only about 1/5 of the original AHSMOG cohort members; who may differ from those who did not participate in this part of the study in ways that may affect estimation of the PM-lung function relationship; (2) these effects were somewhat greater than those observed for a seven pack-year history of past smoking; and (3) about 2/3 of the PM10 data were estimated from TSP.

7.6.2 The Six-Cities and 24-Cities Studies

In the mid-1970s a cohort of white first- and second-grade school children (n = 10,106) in six cities in the eastern U.S. were enrolled in a study to examine both cross-sectional and longitudinal relationships between air pollution and respiratory disease and lung function growth. The mean annual TSP concentrations ranged from 39.3 (Portage, WI) to 114.1

(Steubenville, OH) $\mu g/m^3$, while the corresponding range for sulfates was 5.4 to 18.8 $\mu g/m^3$. Exploring the relationships between pollutant levels in the year preceding the second annual health examination of the children, Ware and colleagues (1986) reported significant relationships between both average PM concentrations (measured as TSP) and sulfates (i.e., the sulfate fraction of TSP) and cough frequency, bronchitis and a composite index of lower respiratory illness. For a 10 $\mu g/m^3$ increase in sulfates, the odds ratios for these three health outcomes were 1.60, 1.68, and 1.57, respectively. Sulfate levels in the 6 cities ranged from 4.4 to 19.3 $\mu g/m^3$. These air pollution – health outcome relationships were observed when the analysis focused on inter-city pollutant differences, but were not supported by analyses within each city over time. No relationship was observed between any of the air pollution metrics and lung function, even when the analysis was restricted to lifetime residents of the six cities.

In a subsequent analysis involving several highly correlated PM metrics (TSP, PM15, PM2.5, and sulfates, measured during 1980-81), all were found to be related to chronic cough, bronchitis, and chest illness reported on health questionnaires (Dockery et al., 1989). Comparing the least and most polluted cities for PM15 (Portage, Wisconsin, and Steubenville, Ohio, respectively), the annual mean concentrations were 20.1 $\mu g/m^3$ and 58.8 $\mu g/m^3$. For PM2.5 the range was 11.8 – 36.7 $\mu g/m^3$, represented by Topeka, Kansas and Steubenville, respectively. Across the range of PM15, the odds ratios for these three health outcomes for all children were 3.7 (95% C.I. = 1.0 – 13.5) for chronic cough, 2.5 (95% CI = 1.1 - 6.1) for bronchitis, and 2.3 (95% CI = 0.8 – 6.7) for chest illness. For sulfates and PM2.5, the odds ratios for these outcomes were approximately doubled; however, unlike the results for chronic cough and bronchitis in relation to PM15, these effect estimates were not statistically significant. There was no association between any pollutant and asthma or persistent wheeze. However, when the analysis was stratified by the presence of asthma or persistent wheezing, the fine particle-related odds ratios for bronchitis and chest illness among those with these conditions were about 60% higher than for the group as a whole, but nevertheless were still not significant. Among the asthmatic and wheezy children, odds ratios for these symptoms in relation to PM15 were at least as high as those for the fine particle metrics, and also were significant for chest illness, and remained so for the nonasthmatic children for the other symptoms. While these results suggest that the combined coarse and fine fractions (measured as PM15) were likely more influential than PM2.5 or sulfates alone in relation to chronic respiratory symptom reporting, the estimates were not statistically distinguishable (i.e., there was substantial overlap between the confidence intervals around the odds ratios for each metric). Finally, as in the earlier report on this cohort, there was no relationship between any PM metric and lung function.

Subsequently, the same group of investigators evaluated the relationships of several air pollutants, including PM10, PM2.1, fine particle sulfate and strong acidity, to respiratory symptoms and lung function in 13,369 white children, aged 8 to 12, in 24 suburban communities throughout the U.S. and Canada (Dockery et al., 1996; Raizenne et al., 1996). Three of the 24 communities were located in California (Livermore, Monterey, and Simi Valley). Particle measurements in each city took place every other day over a one-year period, based on the assumption that this would serve as a reasonably representative surrogate for longer-term exposures; nevertheless this study is essentially cross-sectional in design. Mean PM concentrations over all 24 cities in this study were as follows: PM10 = 23.8 $\mu g/m^3$ (SD=5, range 15.4 – 32.7), PM2.1 = 14.5 $\mu g/m^3$ (SD = 4.2, range 5.8 – 20.7), and sulfates = 4.7 $\mu g/m^3$ (SD = 2.2, range 0.7 – 7.4). Neither PM10 nor PM2.1, *per se*, was associated with any chronic respiratory symptoms. Comparing cities with the highest and lowest annual concentrations, sulfates were associated with at least one episode of bronchitis (OR = 1.65, 95% C.I. = 1.12-2.42) and with any bronchitic symptom (OR = 1.27, 95% C.I. =

1.01-1.61); fine particle strong acidity (which includes sulfates) was linked with bronchitis (OR = 1.66, 95% C.I. = 1.11-2.48). There were no obvious susceptible subgroups within this study population.

Acceptable lung function data were obtained from a subset of 10,251 children in 22 of the 24 communities. All measures of particles were reported to be associated with small, but statistically significant, decrements in several measures of lung function across the ranges of each pollutant. The greatest point estimates of effect were observed for particle strong acidity. For instance, a change in particle strong acidity of 52 nmol/m^3 was associated with the following percentage decrements: forced vital capacity (FVC) = -3.45 (95% C.I. = -4.87 - -2.01), forced expiratory volume in one second (FEV_1) = -3.11 (95% C.I. = -4.62 - -1.58), and peak expiratory flow rate (PEFR) = -3.71 (95% C.I. = -7.10 - -0.20). Still, the estimated lung function differences associated with the range of strong particle acidity could not be statistically distinguished from those related to the other particle metrics. More importantly, because of the cross-sectional nature of this investigation, it is not possible to postulate a causal relationship between any particle metric and long-term decrements in the growth and development of children's respiratory function. This would require a prospective design, such as that employed in the Children's Health Study.

7.6.3 Children's Health Study

Children may be at greater risk from long-term exposures to particles or other air pollutants because the growth and development of the respiratory system may be permanently affected by early environmental insults. Funded by the California Air Resources Board, the Children's Health Study was designed as a 10-year investigation of the impacts of southern California air pollution on lung growth and development and other indices of respiratory health among 3,676 fourth-, seventh-, and tenth-graders in 12 communities, which were chosen to emphasize different long-term air pollution conditions. For data collected in 1986-90, prior to the health data collection efforts, the 24-hr average PM concentration ranged from 28.0 $\mu g/m^3$ in Atascadero and Santa Maria to 84.9 $\mu g/m^3$ in Mira Loma and Riverside. In 1994, the mean 24-hr average PM10 concentration across the 12 communities was 34.8 $\mu g/m^3$ (range = 13.0 $\mu g/m^3$ in Lompoc to 70.7 $\mu g/m^3$ in Mira Loma) (McConnell et al., 1999; Peters et al., 1999a). Although the full 10 years of follow-up data have not been analyzed yet, the initial cross-sectional analysis and some longitudinal results have been published. At enrollment, neither PM10 nor PM2.5 were associated with respiratory illness among the total cohort (ever or current asthma, bronchitis, cough, or wheeze) assessed by questionnaire (Peters et al., 1999a). In contrast, among children with asthma, respiratory symptoms increased with increasing particle levels (McConnell et al., 1999). Specifically, there was about a 40% increase in risk of bronchitis among asthmatics per 19 $\mu g/m^3$ change in PM10 measured over 2-week intervals (OR=1.4, 95% C.I. = 1.1-1.8). Exposure to a 15 $\mu g/m^3$ increment in fine particles resulted in about the same magnitude of increase in risk, which was not statistically significant. Both measures of PM were also associated with at least a doubling of risk of phlegm in asthmatic children. Acid vapors and NO_2 were also associated with respiratory symptoms in asthmatic children. However, because all four (PM10, PM2.5, NO_2, and acid vapor) were highly correlated, it is not possible to definitively attribute these effects to any single pollutant (McConnell et al., 1999).

In another cross-sectional analysis of the Children's Health Study, both PM10 and PM2.5, as well as NO_2, were significantly associated with decreased lung function (forced vital capacity [FVC], forced expiratory volume in one second [FEV_1], and maximal mid-expiratory flow [MMEF]), especially in girls who spent more time outdoors (Peters et al., 1999b). Recently these results were supported in an analysis of lung function growth over a four-year period

(Gauderman et al., 2000). Examining the data from a sample of children who were fourth-graders at enrollment, the investigators found statistically significant effects on lung function growth associated with PM10, PM2.5, PM10-2.5 (coarse particles), NO_2, and inorganic acid vapors. The effects were more pronounced for tests measuring airflow at low lung volumes, especially for children spending more time outdoors. However, unlike the cross-sectional results, there were no differences observed by gender. Although the effects on the children who were seventh- and tenth-graders at enrollment were generally also negative, these were not statistically significant, in part because the sample sizes in the higher grades were markedly smaller. As with the cross-sectional symptom data, the independent effects of the different pollutants cannot be assessed because of high inter-pollutant correlations.

Although data on sulfate concentrations have been collected as part of the Children's Health Study, no analyses examining potential independent effects of this component of PM2.5 have been published. According to ARB staff, such analyses will be conducted during the next few years.

7.6.4 The SAPALDIA Study

The Swiss Study on Air Pollution and Lung Disease in Adults (SAPALDIA) examined the long-term effects of air pollution exposure in a cross-sectional study of 9,651 adults residing in eight areas in Switzerland in 1991. Eligibility for the study was conditional on having lived in the same area for at least three years. PM measurements used in the analysis were taken over a 1-year period (1991 for TSP, and 1993 for PM10), on the assumption that air pollution concentrations had not changed significantly over the proceeding several years. Significant associations were observed between chronic symptoms (chronic phlegm, chronic cough, breathlessness at rest during the day or at night, and dyspnea on exertion) and the pollutant metrics TSP, PM10 and NO_2 (Zemp et al., 1999). These associations were strongest for PM10; the investigators estimated that an increase of 10 $\mu g/m^3$ PM10 (within the observed range across cities of 10.1 – 33.4 $\mu g/m^3$), would correspond to increases in risk among never-smokers of 30% for chronic phlegm (OR=1.30, 95% C.I. = 1.04-1.63), 41% for breathlessness during the day (OR=1.41, 95% C.I. = 1.13-1.76), and 23% for dyspnea on exertion (OR = 1.23, 95% C.I. = 1.09-1.39). Nevertheless, the roles of PM10 versus NO_2 in the observed associations could not be ascertained, as NO_2 concentrations were strongly correlated with PM10 levels (r = 0.91).

The SAPALDIA investigators also examined lung function (FEV_1 and FVC) in study participants in relation to several air pollutants, controlling for age, sex, height, weight, atopy, educational level, nationality, smoking status (never, ever, and current), workplace exposures, residential gas stove, serious respiratory infection before age 5, and other potentially influential covariates (Ackermann-Liebrich et al., 1997). Statistically significant decrements in both indices of lung function were found in relation to annual mean levels of PM10, sulfur dioxide, and nitrogen dioxide, with the strongest effects being related to PM10 (-3.4% for FVC and –1.6% for FEV1 in healthy never-smokers, per 10 $\mu g/m^3$ annual average PM10, $p<0.001$ for both estimates). The mean PM10 concentration in this study (measured only in 1993) was 21.2 $\mu g/m^3$ (SD = 7.4), with a range of 10.1 – 33.4. Similar, but slightly smaller, estimates were found for past and current smokers. As with the respiratory symptom analysis, however, the strong pollutant inter-correlations made it impossible to disentangle the effects of the various pollutants ($r_{PM10,SO2} = 0.93$; $r_{PM10,\ NO2} = .91$, $r_{SO2,NO2} = 0.86$). Thus, they concluded that the principal source of all three pollutants, fossil fuel combustion, was associated with the decrements in lung function.

7.6.5 Summary

In summary, the evidence of PM effects in these studies of morbidity in relation to chronic exposures is not as consistent as for mortality. In several studies, the various PM metrics are highly inter-correlated, or co-varied with gaseous pollutants, so that it was not possible to attribute the effects observed to any single pollutant or to a specific mix of pollutants (e.g., the Six-Cities, Children's Health, and SAPALDIA studies). In studies examining effects of exposure to different PM metrics, in some cases the point estimates of effect were greater for those metrics encompassing the coarse fraction (e.g., Dockery et al., 1989), and in some cases the reverse was true. Overall, there is some, albeit weak, evidence of a PM-related effect on chronic morbidity, as measured by chronic respiratory symptoms and lung function. However, it is not possible, based on current evidence, to identify which size cuts or specific constituents are likely to be most influential.

7.7 Susceptible Subgroups at Risk for Mortality

7.7.1 By Disease Status

Pre-existing cardiovascular disease is clearly a risk factor for PM-related mortality. Many of the time-series studies, and both the ACS and Harvard Six-City chronic exposure studies, report statistically significant associations for cardiovascular-specific mortality (for example, Samet et al., 2000a; Ostro et al., 2000; Fairley, 1999; Schwartz, 1993). When compared with all-cause mortality, the cardiovascular-specific mortality typically (but not always) generates larger and more certain effect estimates for PM. These conditions might be further exacerbated by pre-existing respiratory disease. Several mortality studies of acute air pollution exposure provide evidence to identify the most likely sensitive subgroups among adults. For example, Schwartz (1994b) reported that respiratory conditions were more likely to be contributing causes of death on high versus low PM days. Thus, air pollution was associated with increased deaths from respiratory conditions and increased deaths from other causes with respiratory conditions as a contributing factor. In a study of hospital admissions in Cook County, Zanobetti and Schwartz (2000b) found that acute bronchitis and pneumonia increased the risk for admission to hospital with cardiovascular disease. Finally, in a daily mortality study in Montreal (Canada), Goldberg et al. (2000) found that the association between PM and mortality was elevated among those with acute lower respiratory disease, coronary artery disease, congestive heart failure, and any cardiovascular disease. No risk elevation was observed for those with acute upper respiratory disease, airways disease (which was defined to include chronic bronchitis, emphysema, asthma, and bronchiectasis), acute coronary artery disease (i.e., acute myocardial infarction, and other acute and subacute forms of chronic ischemic heart disease), hypertension or cerebrovascular disease (i.e., stroke). Taken together, these studies suggest that concurrent lower respiratory infections and subsets of cardiovascular disease may be precursors to death associated with PM.

7.7.2 By Socioeconomic Status

Several mortality studies have examined whether socioeconomic status (SES) and related factors such as education and race/ethnicity affect the magnitude of PM-mortality associations. These studies help address the question of whether factors linked with poverty or educational attainment render individuals more susceptible to the adverse effects of exposure to air pollution. To date the findings have been mixed. The prospective cohort studies investigating the potential impacts of longer-term exposure appear to find consistent effect modification by education, whereas the acute exposure studies do not demonstrate much, if any, modification of these relationships.

In their re-examination of the American Cancer Society (ACS) data set originally analyzed by Pope et al. (1995), Krewski et al. (2000) conducted an exhaustive set of sensitivity analyses. They considered a wide range of alternative specifications, ecological variables, corrections for spatial autocorrelation, interactions, adjustment for time-varying parameters, and measures of occupational exposure, smoking, and physical activity. Their findings corroborated those of the original study.

However, the relative risk (RR) estimates from the prospective cohort studies vary significantly when the analysis was stratified by educational attainment (Table 7.5). For those with a less than high school education, the relative risk (RR) associated with an inter-quartile change in the annual average fine particle concentration was 1.35 (95% C.I. = 1.17 – 1.56), while for those with more than a high school education, the RR = 1.06 (95% C.I. = 0.95- 1.17). This lower risk associated with more education was also observed in the education-stratified re-analysis of the Dockery et al. (1993) study (Krewski et al., 2000). The lack of an association among more well-educated individuals may indicate that better nutrition and access to health care (or some other variables correlated with educational attainment) may be important co-factors in air pollution-associated mortality. The effect of SES did not appear to be confounded by occupational exposures in these cohorts. For example, among the groups with either low or high occupational exposures, higher educational attainment was associated with lower risks from air pollution. Among individuals with lower educational attainment, poverty, poor nutrition, and less access to medical resources are all more common. Anecdotally, lower SES is also likely to be associated with residences closer to mobile and stationary sources of pollution. Therefore, it is possible that SES is simply associated with higher exposure to existing sources, rather than an effect modifier.

In a third prospective cohort study (of Seventh Day Adventists in California), McDonnell et al (2000) analyzed a subset located close to airports, in order to utilize airport visibility as a surrogate measures of PM2.5. For the population as a whole, no association was observed between alternative measures of PM (fine, coarse or PM10) and either all cause mortality or non-cancer respiratory mortality. Similarly, no association was apparent for the male cohort. This group was then further disaggregated by other subsets including individuals who were: past smokers, exposed occupationally, exposed to ETS, with a history of cardiovascular or respiratory disease, using antioxidant pills, living in high-density areas, and not using alcohol. The largest observed effect, which was statistically significant, was among those living in high housing density, which is often associated with low SES.

There is some, albeit fragmentary, evidence of effect modification of the PM-mortality relationship by income or education. For example, Zanobetti and Schwartz (2000) tested for effect modification in the four largest cities with daily PM10 data during the study period of 1986 – 1993 (Chicago, Detroit, Minneapolis-St.Paul, Pittsburgh). They used individual-level educational status from the death records of the National Center for Health Statistics. In three of the four cities, the PM10 effect for the cohort members with less than 12 years of education was larger than that for those with more than 12 years of education. In two of the cities, the PM effect for those in the low-education group was more than twice the other cohort. Thus, there was weak evidence of effect modification by education. In contrast, in a study of air pollution and mortality in 10 U.S. cities, Schwartz (2000a) examined whether the city-specific mortality effect was modified by several city-wide factors. No effect modification of the pollution effect was found from unemployment, living in poverty, college degree or the proportion of the population that is nonwhite, although sample size limited the ability for detection. Samet et al. (2000a) tested for effect modification of the PM10-mortality association among the 90 cities used in the study. Using aggregate (city-wide) statistics, they tested for potential modification using local SES-related variables, including household income, percent

of the population having less than a high school education, percent using public transit, and percent unemployed. None of these factors helped explain the city-specific pollution effects. However, the variable representing the percent of the population having less than a high school education had a moderate (but still not statistically significant) association with the regression coefficients.

The evidence to date, therefore, suggests that there may be a greater effect of PM among individuals from lower SES groups, although the actual risk factors are unknown. Candidate risk factors include poor nutrition, lower access to and use of health care, and higher air pollution exposures due to location of residences near PM sources such as freeways and industrial facilities.

7.7.3 By Age

7.7.3.1 The Elderly

Existing evidence suggests that most of the more severe effects of PM are likely to be experienced by elderly people with pre-existing heart or lung disease. For example, when the acute exposure mortality studies have disaggregated the sample by age, the elderly subsample typically exhibits stronger associations and larger effect sizes. In some extreme cases, statistically significant effects are observed only for the elderly subset (Goldberg et al., 2000; Kelsall et al., 1997). As summarized in Table 7.6, PM has, in general, a disproportionate effect on the elderly. For example, a study in Brisbane, Australia (Simpson et al., 1997) found that 81% of all mortality occurred in the age group above 65, but 90% of the PM-related mortality occurred in this group. Likewise, in Santiago, Chile (Ostro et al., 1996) the rates are 65 and 79%, respectively. Thus, a large share, but not all, of the acute-exposure mortality occurs within the elderly population.

7.7.3.2 Infants and Children

While the elderly may dominate the potential population at risk, several recent cross-sectional and time-series studies have reported associations between ambient PM and neonatal or infant mortality, low birthweight or higher rates of prematurity. For example, in Rio de Janeiro (Penna and Duchiage, 1991) and the United States (Woodruff et al., 1997), cross-sectional associations have been reported between measures of PM and neonatal or infant mortality. Woodruff et al. (1997) studied a cohort of four million infants born between 1989 and 1991, who were studied using data from the National Center for Health Statistics. Infants were assigned three different PM10 exposure intervals based on metropolitan area-wide data averaged over the first 2 postnatal months. The mean PM10 was 31 $\mu g/m^3$. Logistic regression was used to examine whether there was an association between early neonatal exposure and total or cause-specific mortality, after controlling for other demographic and environmental factors. Associations were found between higher PM10 exposure and both all-cause and respiratory-specific mortality.

Another study (Dejmek et al., 1999) evaluated the impacts of PM2.5 and PM10 on intrauterine growth retardation (IUGR) in the highly polluted Teplice District in the Czech Republic. Again, three different exposure intervals were determined for several pollutants (PM, nitrogen dioxide and sulfur dioxide) for each month of gestation. Data analysis found no effect from nitrogen dioxide, but PM10 and sulfur dioxide in early pregnancy were associated with IUGR, after controlling for several potential confounders. Both PM10 and PM2.5 (which were highly correlated in this study) were associated with the likelihood of an IUGR birth, defined as one where the birth weight fell below the 10^{th} percentile by gender and age for live births in the Czech Republic. These results suggest that exposure to PM in Teplice (which includes

PM2.5, PM10, sulfates, acid aerosols and PAHs) early in pregnancy may impact subsequent fetal growth and development.

Bobak and Leon (1998) conducted a matched case-control study of all births registered in the Czech Republic from 1989 to 1991, which were linked to death records. A logistic model was used to estimate the effects of PM on the risk of death, after controlling for socioeconomic status, birth weight and length, and gestational age. An association was found between PM and post-neonatal respiratory mortality. Bobek (2000) used a somewhat similar database of live births registered in the Czech Republic in 1990-1991 to examine associations between air pollution and both low birth rate and prematurity. The birth outcomes were linked with pollution data on TSP, sulfur dioxide and nitrogen dioxide in the 67 of 85 districts (about 85% of all births) for which data were available. Outcomes studied included the likelihood of lower birth weight (<2,500 g), prematurity (< 37 weeks of gestation) and IUGR (< 10^{th} percentile of birth weight for gestational age and sex). The analysis controlled for sex, parity, maternal age group, education, marital status, nationality and month of birth. Associations were found between TSP (median concentration = 72 $\mu g/m^3$) and both low birth weight and prematurity, but not with IUGR. The association with TSP and low birth weight appeared to be explained by low gestational age.

In both the cross-sectional and case-control study designs, it may be difficult to separate the effects of pollution from other factors such as poverty, exposure patterns (e.g,, in the higher pollution areas people may spend more time outside or live closer to highways), and diet. However, daily time-series studies have also reported associations between changes in PM and infant or child mortality in Mexico City (Loomis et al., 1999) and Bangkok (Ostro et al., 1999a). The statistical models used in these studies were similar to those used in the adult mortality studies of acute exposure – general additive Poisson models, controlling for time, season and weather. In Mexico City, 3- to 5-day lags in PM2.5 (mean = 27 $\mu g/m^3$) were associated with infant (< 1 year) mortality. Likewise, in Bangkok, lags of 2 or 3 days of PM10 (mean = 65 $\mu g/m^3$) were associated with child (< 5 years) mortality. These two studies suggest about a 2 to 4% increase in daily infant mortality per 10 $\mu g/m^3$ PM10. In both of the cities, however, the personal exposure to PM is likely to be much greater than in the U.S. due to factors such as weather, poverty, time spent outdoors, and housing ventilation. In addition, differences in prenatal maternal health status and early postnatal infant diet may make it difficult to extrapolate these findings to California.

Finally, Ritz et al. (2000) reported associations between PM and both low birth weight and premature delivery among a cohort of 98,000 neonates born in Southern California between 1989 and 1993. Prematurity was defined as a birth occurring at less than 37 weeks of gestation. Seventeen monitoring stations throughout the Los Angeles air basin had data for at least four pollutants of interest, including PM10, carbon monoxide, nitrogen dioxide and ozone; only 8 of the stations had PM10 data. Only births for women living within 2 miles of a monitoring station were included in the analysis. Pollution exposures were averaged over several distinct periods, such as 1, 2, 4, 6, 8, 12, and 26 weeks before birth and the entire pregnancy, as well as averages over the first and second months of pregnancy. Several known risk factors were controlled for, including maternal age, race, education, parity, sex of the infant. However, data were not available for maternal smoking or exposure to ETS, marital status, maternal height, and pregnancy weight gain. Ultimately, the strongest association was found between PM10 averaged over the 6 weeks prior to birth and the likelihood of pre-term birth.

7.7.4 Summary

Taken together, the evidence to date suggests that exposure to PM is likely to have a disproportionate effect on the elderly, and possibly on children and infants. The impacts on the elderly have been observed in both the mortality and the hospitalization studies. In the latter, associations were found between PM10 and hospitalization for both cardiovascular disease and respiratory diseases, including COPD and bronchitis. These outcomes are observed primarily in the elderly, and many of the studies restricted the sample to those above age 65. It may be premature to generalize the findings of the effects of PM exposure on infants. Many of the studies were cross-sectional in nature, making it more difficult to attribute the effect to a single factor. On the other hand, the time-series studies were undertaken outside of the U.S., where the pollution concentrations, exposure conditions and underlying socioeconomic factors may be very different from that in the U.S. Besides predicting mortality, several studies have reported associations between exposure to PM and low birth weight, prematurity, and IUGR.

7.8 Biological Mechanisms

7.8.1 Overview

Until recently, there was no clear mechanistic explanation for the observed epidemiological findings of mortality and morbidity following acute or subacute exposure to ambient particles, especially those findings referable to the cardiovascular system. However, within the past few years epidemiological and controlled exposure studies in human subjects, as well as some toxicological investigations, have provided evidence of several biologically plausible mechanisms that may underlie some of the serious adverse effects observed in the time-series investigations. The initial target organ affected by exposure to particles is the lung, though small particles can penetrate into the blood and be detected in the systemic circulation within minutes of inhalation (Nemmar et al., 2001). Within the lung, effects have been observed in both the conducting airways and the gas-exchange zone, both of which may result in local and systemic effects. In epidemiological studies examining the relationships between PM pollution and mortality, in particular, effects have often involved the cardiovascular system as well. Much of this section will focus on recent research suggesting mechanisms by which systemic effects, particularly those affecting the heart, may occur.

The basic pathophysiological models of PM-related health impacts begin with deposition of PM in the airways and the alveoli, eliciting an inflammatory response, and potentially affecting pulmonary defenses against infection. Inflammation is a stereotyped biological response to injury or infection and, although necessary in principle for the defense of the organism's physiological integrity, can also result in amplification of injury, both locally and systemically. A variety of cell types in the lung (e.g., alveolar macrophages and epithelial cells) may respond to the presence of particles by secreting chemical messengers (cytokines and chemokines), which in turn can attract inflammatory cells to the lungs from the bone marrow and other parts of the body. Particles may also adversely affect the ability of macrophages to protect the lung against inhaled micro-organisms, which could result in enhanced susceptibility to infection. Programmed cell death (apoptosis) may be induced in both epithelial cells and macrophages by particles, further reducing native defenses against environmental stresses. Inflammation of the bronchi and bronchioles is associated with airway hyperresponsiveness, represented by an increased propensity of smooth muscle cells of the airways to constrict in response to irritants, cold air, pharmacological spasmogens, and other agents.

This series of acute responses may also involve effects on the autonomic nervous system and the composition of the blood. Chronic lung diseases, including asthma, emphysema, and

chronic bronchitis, all involve ongoing, unresolved inflammation in the lung. Additional inflammatory stimuli in the lungs could exacerbate chronic lung disease, resulting in bronchoconstriction and respiratory symptoms, as well as reduced oxygenation of the blood. In addition, pre-existing pulmonary inflammation may facilitate PM-induced release of pro-inflammatory mediators, resulting in additional pulmonary inflammation and systemic, including cardiac, effects.

7.8.2 Pulmonary and Systemic Inflammation

Reports in humans and experimental animals suggest that inhalation of particles from diverse sources can cause pulmonary and systemic inflammatory responses. In a variety of *in vivo* animal and *in vitro* experimental models (Costa and Dreher, 1997; Kennedy, et al., 1998; Brain, et al., 1998; Li et al., 1996), exposures to high concentrations of PM have been found to cause lung inflammation, cell and tissue injury, and changes in cell populations. In many cases, toxicological studies involving high-level exposures *in vitro* or by intra-tracheal instillation or inhalation indicate that the presence of soluble transition metals (e.g., iron, vanadium, nickel) enhances inflammatory responses (Kodavanti et al., 1997, 1999; Monn and Becker, 1999; Costa and Dreher, 1997; Li et al., 1997). These metals may generate localized oxidative stress through the formation of oxygen-based free radicals, such as the potent hydroxyl radical (Donaldson et al., 1997). The injury caused by oxidative stress may lead to a decrease in epithelial integrity, resulting in enhanced transfer of particles into the lung interstitium. The presence of particle-associated metals is not, however, a *sine qua non* for inflammation to take place. Ultrafine carbon black particles (i.e., particles of aerodynamic diameter less than 100 nm or 0.1 μm) appear to cause markedly greater inflammation than fine particles in experimental settings; these effects of ultrafine particles are not mediated by soluble metals or iron at the particle surface (Brown et al., 2000). Moreover, on a mass basis, ultrafine carbon black particles exert a greater effect than fine particles *in vitro* on alveolar macrophage function (Renwick et al., 2001), which could, in theory, affect the host's ability to clear other particles, including infectious micro-organisms.

PM-associated organic compounds such as polycyclic aromatic hydrocarbons (PAHs) also appear to exert toxic effects in the lung via oxidative stress. A series of experiments using diesel exhaust particles (DEP) demonstrates the induction of reactive oxygen species (ROS, including hydrogen peroxide and superoxide) by both lung macrophages and epithelial cells (Nel et al., 2001). Generation of such oxidant stress can activate specific transcription factors, including nuclear factor κB and activator protein-1, which can upregulate the expression of genes for cytokines, chemokines, and other pro-inflammatory mediators. DEPs or organic extracts of DEPs can also, through oxidant effects on mitochondria, induce programmed cell death (apoptosis) or necrosis of macrophages and respiratory epithelial cells (Nel et al., 2001). Moribund macrophages release additional ROS in their immediate environments, amplifying the oxidative stress and, in addition, would be unable to engulf and kill infectious micro-organisms. Apoptosis of respiratory epithelial cells could lead to a loss of integrity of the lining of the airways, which may facilitate airway hyperresponsiveness and exacerbation of asthma or other conditions involving airway inflammation. Generation of oxidant stress has also been demonstrated both *in vivo* and *in vitro* after exposure to concentrated, resuspended PM2.5 and ultrafine carbon black (Shukla et al., 2000).

Although there has been little toxicological work examining potential impacts of coarse versus fine particles, some recent literature indicates that the coarse fraction may be capable of eliciting greater pro-inflammatory effects than the fine fraction, due at least to metals and endotoxin in the coarse fraction (Monn and Becker 1999; Soukoup et al., 2001). Endotoxin is a generic name for an essential component of gram-negative bacterial cell walls, and is nearly ubiquitous in soils. Exposure of humans to endotoxin in largely occupational settings has

resulted in increased lung inflammation, enhanced airway responsiveness, increases in systemic immune cell populations, and decrements in lung function (Michel et al., 1997; Vogelzang et al., 1998; Zock et al., 1998). Monn and Becker (1999) demonstrated the importance of endotoxin associated with the coarse particle fraction (PM10-PM2.5) in the induction of pro-inflammatory cytokines, such as interleukin-6. In these *in vitro* studies, coarse fraction PM induced cytokine levels 50 times higher than those seen with the soluble fractions of coarse PM or fine-mode particles. Kleinman et al. (1995) demonstrated that lung permeability, a measure of cell damage and inflammation, was increased by coarse fraction road dust exposure in a dose-dependent fashion. While the relevance of such work to human responses to ambient PM remains to be established, it is clear that different size cuts of PM (coarse, fine and ultrafine) of PM10 can deposit throughout the airways (see Section 7.1), and have the potential to elicit intrapulmonary inflammation and compromise the functional abilities of alveolar macrophages.

The intrapulmonary responses elicited by PM may be due in part to neurogenic inflammation. Sensory neurons in contact with irritant particles (e.g., within the conducting airways) can be stimulated to release neuropeptides (e.g., substance P, calcitonin gene related peptide, neurokinin A and others), which can initiate airway inflammatory events, including release of cytokines, vasodilation, and mucus secretion. Neuropeptides act on a variety of cell types within the lung, including epithelial and smooth muscle cells (resulting in modulation of inflammation and airway hyperresponsiveness), as well as immune cells (polymorphonuclear cells or PMNs, lymphocytes, eosinophils, and others), which can amplify the inflammatory response. Recent *in vitro* experiments indicate that specific irritant (capsaicin or vanilloid) receptors on neurons are necessary for PM-related neurogenic inflammation to occur, as evidenced by responses to several types of particles, including ambient particles collected from St. Louis and Ottawa, coal fly ash, residual oil fly ash, and particles from the eruption of Mt. St. Helens (Veronesi et al., 2000).

Several controlled exposure studies in humans clearly demonstrate that particle inhalation evokes an inflammatory response. Salvi et al. (1999) exposed 15 healthy human adult volunteers to either air or diesel exhaust (PM10 concentration = 300 $\mu g/m^3$) for an hour each, at least 3 weeks apart, and examined inflammatory responses 6-hr post-exposure in bronchial washings, bronchoalveolar lavage fluid, bronchial biopsies, and in peripheral blood samples. They observed a vigorous inflammatory response in the samples obtained from the lung, including significantly increased numbers of PMNs, T- and B-lymphocytes, mast cells, inflammatory mediators (histamine and fibronectin), as well as several adhesion molecules that facilitate the passage of inflammatory cells from the circulation into the airways. In the blood samples, they reported increased PMNs and platelets (cells involved in the initial formation of blood clots), suggesting that the diesel exposure stimulated the bone marrow to release these cells into the circulation and then to the airways.

Subsequently, the same group of investigators reported that this diesel exposure protocol also resulted in increased intra-airway transcription of messenger RNA for interleukin-8 (IL-8), a protein that attracts PMNs to sites of injury (Salvi et al., 2000). In addition, they detected increased production of IL-8 and another protein (GRO-α), both of which promote inflammation, in the subjects' airways. Another laboratory (Nightingale et al., 2000) also reported evidence of airway inflammation following a different experimental protocol in 10 healthy adult volunteers (involving 2-hr exposures to 200 $\mu g/m^3$ of re-suspended diesel exhaust particles, with different timing and methods of obtaining intra-airway specimens). Though this group found no increases in three mediators of inflammation in the subjects' blood, they did report an increase in exhaled carbon monoxide after diesel exhaust exposure, suggesting the presence of oxidative stress in the lung. Though their findings are not entirely

consistent with those of Salvi et al. (1999, 2000), some of the discrepancies are likely due to differences in the study designs and methods. For instance, some of the discordance may be due to the timing of sample collection; inflammatory responses follow a consistent succession of events, with increases in different cytokines and cell types occurring sequentially. These events begin within hours of the initial exposure, which could help explain the short time lag between exposure and outcome observed in some time-series studies (Nordenhäll et al 2000).

Taken together, these publications suggest a potential pathway by which particles might increase airway inflammation and provoke exacerbations of chronic respiratory disease such as asthma. However, these data should be interpreted cautiously. First, the exposure concentrations were relatively high: ambient particle levels rarely reach 200 - 300 $\mu g/m^3$ in the U.S., though this range is not uncommon in some of the larger cities in the developing world. In addition, diesel exhaust exposures may not be representative of PM generally, and are well recognized to enhance allergic inflammation (Nel et al., 1998). However, in some cities outside the U.S. (such as London, UK, or Santiago, Chile), diesel exhaust particles comprise the majority of small particles (QUARG 1993; Sandoval et al., 1985). Moreover, other particles administered in high doses (e.g., residual oil fly ash) are capable of amplification of the allergic response in experimental animals (Gavett et al., 1999). Finally, the Salvi et al. studies involved whole diesel exhaust, which also contains oxidant gases known to enhance intra-pulmonary inflammation.

In a controlled study using particles potentially more representative of those to which the general population is exposed, Ghio et al. (2000) reported evidence of mild airway inflammation, without concomitant lung injury. In this study the investigators used concentrated ambient particles (CAPs) collected in the immediate vicinity of the Human Studies Facility of the U.S. EPA in Chapel Hill, NC. The investigators exposed 38 healthy adults once to either clean air (n = 8) or CAPS (n = 30) for 2 hours, with intermittent exercise. The CAPS exposures ranged from 23.1 to 311.1 $\mu g/m^3$ of PM2.5, with a mean concentration of 120.5 $\mu g/m^3$. Technical limitations of the concentrator restricted the range of particles collected to those with diameters between 0.1 and 2.5 μm. As reported in the controlled diesel exposure studies (discussed above), they found an influx of PMNs into the airways (an approximately 3.7-fold increase in bronchial washings and 6.2-fold increase in bronchoalveolar washings obtained 18 hr post-exposure), comparable to what has been observed among individuals exposed to low concentrations of ozone for several hours. However, they found no increase in indicators of lung injury or in the concentrations of a variety of pro-inflammatory mediators (IL-8, IL-6, fibronectin, and others) in the lung lavage fluid. Moreover, most of the blood parameters analyzed showed no exposure-related changes (hemoglobin, hematocrit, red blood cell count, PMNs, lymphocytes, other white blood cells [monocytes], platelets, ferritin [an iron transport protein that can increase during the early phases of an inflammatory reaction], or blood viscosity). However, fibrinogen, a key constituent involved in blood clotting, was elevated by the CAPS exposures relative to clean air ($p = 0.009$), with no obvious dose-dependence. Thus, while not entirely consistent with the diesel exhaust controlled exposure studies discussed in preceding paragraphs (which may be due in part to differences in experimental protocol), this CAPS study suggests that exposures to ambient particles in healthy humans can result in a mild pulmonary inflammatory response. Though the exposure concentration was higher than what would ordinarily be encountered in the U.S., the cumulative particle exposure experienced by most of the subjects in this experiment would be lower than 24-hr PM exposures in many urban areas.

Tan et al. (2000) obtained venous blood samples at weekly intervals from 30 military recruits in Singapore who followed standardized outdoor activities throughout the Southeast Asia

haze episode of 1997 resulting from wildfires in Indonesia. Measures of immature PMNs in the subjects' blood were analyzed in relation to daily measures of several pollutants (including 24-hr PM10) monitored during and after the haze episode, which lasted for about 5 weeks. During the episode the mean PM10 concentration was 125.4 $\mu g/m^3$, while afterwards it was about 40.0 $\mu g/m^3$. Tan et al. (2000) found the strongest relationship between same-day PM10 and increased immature PMNs in the circulation, though there was also a statistically significant relationship with a one-day lag. Although not sufficient to establish a cause-and-effect relationship, these results suggest an immediate stimulation of the bone marrow from inhalation of smoke containing high levels of particles, resulting in the early ejection of immature PMNs into the circulation.

In a subsequent experiment in which rabbits had 5 mg of PM10 (previously collected in Ottawa, Canada) instilled intrapharyngeally twice a week for three weeks, the same laboratory found that repeated PM exposure increased the production of PMNs in the bone marrow and accelerated their release into the circulation (Mukae et al., 2001). The PM10 exposure resulted in diffuse inflammation of the lungs, with particles present in alveolar macrophages, lung epithelial cells (Type II pneumocytes), and in the airway walls. The effects on PMN production in bone marrow and release of immature cells into the blood were associated with the numbers of particles ingested by alveolar macrophages. Also, for purposes of comparison, the investigators found that a higher percentage of human alveolar macrophages, obtained from lung sections removed from both smokers and nonsmokers with small lung tumors, contained fewer particles than those taken from the experimental rabbits (Mukae et al., 2001).

For individuals with chronic lung disease, such as asthma or COPD, such pro-inflammatory effects may result in exacerbation of disease. PM effects on alveolar macrophage function may also compromise one of the principal pulmonary defenses against infection (Renwick et al. 2001). The latter may also represent an important pathway for worsening of both asthma and COPD, as serious exacerbation of both conditions is often related to respiratory infection.

Taken together, these data suggest that inhalation of different sources of particles can initiate inflammatory events in human lungs, with some evidence of systemic impacts, notably stimulation of bone marrow to accelerate production of inflammatory cells to respond to the pulmonary insult. As discussed below, changes in the composition of the blood may also result from these effects, with potentially serious effects on individuals with cardiovascular disease.

7.8.3 Effects on the Circulation and Cardiac Events

Several years ago, Seaton (1995) proposed that exposure to ultrafine particles might induce alveolar inflammation, which could lead to exacerbation of pre-existing lung disease and increased blood coagulability. Increased blood coagulability could in turn lead to acute cardiovascular events, notably myocardial infarctions, by the formation of blood clots (thrombi) in compromised coronary arteries, or through the formation of such thrombi in other sites, which subsequently travel through the circulation to the coronary arteries. Research during the past decade has demonstrated that thrombus formation is the critical event in many patients suffering an acute coronary event (Rosito et al. 1996). As described above, several studies of controlled exposures to particles demonstrate increases in both cellular and biochemical markers of inflammation in the lung (Salvi et al. 1999, 2000; Nightingale et al. 2000; Ghio et al. 2000). This observation is subject to the caveat that three of these four studies involved diesel exhaust particles, which may not necessarily be representative of ambient PM generally. The Ghio et al. (2000) study also noted a PM-related increase in fibrinogen, a key component in blood coagulation). Fibrinogen concentrations have been

reported to be elevated in cigarette smokers and individuals exposed to cigarette smoke, which is well recognized as a risk factor for cardiovascular disease (Sato et al. 1996; Iso et al. 1996). At least one study of rats exposed to residual oil fly ash particles at a high dose level (8.3 mg/kg by intratracheal instillation) also found an increase in the animals' blood fibrinogen levels (Gardner et al. 2000). Plasma viscosity was also elevated in these animals, but not significantly so. Some recent epidemiological data suggest potential effects of particulate air pollution on blood coagulation (Peters et al. 1997, Seaton et al. 1999). Recently, PM pollution has also been linked with the onset of myocardial infarction (Peters et al., 2001a). While the existing evidence is still somewhat sparse and is not completely consistent, plausible mechanisms for the time-series results regarding cardiovascular morbidity and mortality are beginning to emerge.

Using data collected as part of a large cross-sectional study of cardiovascular risk factors in southern Germany (MONICA -- MONItoring of trends and determinants in CArdiovascular disease), Peters et al. (1997) analyzed blood viscosity in relation to a 13-day air pollution episode that occurred in January 1985. During the episode, TSP and sulfur dioxide were markedly elevated. The investigators found that, although the distributions of viscosity had not shifted during the episode, there was a tendency (among some of the participants) towards higher values on episode days. During the air pollution episode, the risk of having blood (or strictly speaking, plasma) viscosity above the 95th percentile [determined for the whole study, including before and after the episode] was increased in both genders (OR = 3.62, 95% CI = 1.61-8.13 for men, and OR = 2.26, 95% CI = 0.97-5.26 for women). Odds ratios for increased plasma viscosity related to a 100 $\mu g/m^3$ increment in TSP concentration were also elevated for both men and women, but were not statistically significant. Blood viscosity has been associated with severity of cardiovascular disease (Junker et al. 1998). Moreover, subjects with elevated plasma viscosity also tended to have increased heart rates as well, suggesting multiple pathways of elevated cardiovascular risk (Peters et al. 2000b). Fibrinogen, one of the principal proteins involved in the determination of blood viscosity, is well established as an important independent risk factor for myocardial infarction and stroke (Yarnell et al. 1991; Ernst et al. 1993). However, fibrinogen was not specifically assayed in this investigation.

In a subset of the German MONICA study population, consisting of 631 randomly selected healthy men aged 45 to 64 years, the investigators examined C-reactive protein concentrations in blood obtained during the initial cross-sectional study (1984-85) and again three years later (Peters et al. 2001b). C-reactive protein (CRP) is a sensitive indicator of infection, injury, and inflammation, and has been linked with increased risks of both incidence and exacerbation of cardiovascular disease (Haverkate et al. 1997; Rifai 2001). CRP levels were elevated during the 1985 air pollution episode, with the strongest effects related to TSP. In multivariate regression analyses, the odds of having elevated CRP above the 95th percentile (for the entire study) were increased by $\approx$ 50% for same-day TSP (31 $\mu g/m^3$ inter-quartile range), to $\approx$ 75% for a five-day TSP average (26 $\mu g/m^3$ inter-quartile range). These increases were unchanged even after deletion of the 1985 episode days, indicating that acute and subacute effects could be observed even at normal ambient PM levels: the mean TSP concentrations during the two study periods were 54 $\mu g/m^3$ in 1984/85 and 47.8 $\mu g/m^3$ in 1987-88.

In a large, representative cross-sectional sample of the United States population, Schwartz (2001b) found that ambient PM10 was associated with elevated blood levels of several cardiovascular risk factors. Schwartz (2001b) examined local PM10 concentrations either the same day or the day before an extensive questionnaire and physical examination (including obtaining venous blood samples) were administered to approximately 20,000 individuals in 44 communities as part of the Third National Health and Nutrition Examination Survey. In single

pollutant models, controlling for age, race, sex, body mass index, and cigarette smoking, PM10 concentrations were significantly associated with serum fibrinogen levels, platelet counts, and white blood cell counts. Platelets and fibrinogen were also associated with NO_2, while WBC counts were associated with SO_2, and none of the three blood markers were associated with ozone. In multi-pollutant models, only the coefficients linking PM10 and these cardiovascular risk factors remained significant. Schwartz undertook extensive sensitivity analyses, examining the potential impacts of social factors (poverty, educational attainment, household size), other exposures (environmental tobacco smoke, serum cotinine [a biomarker of exposure to tobacco smoke], use of a wood stove, fireplace, or gas stove), dietary influences (serum vitamin C, intake of fish, shellfish, saturated fat, caffeine, and alcohol), as well as other cardiovascular risk factors (systolic blood pressure, total serum cholesterol and high density lipoprotein levels). The associations between PM10 and fibrinogen, platelet counts, and WBC counts remained robust to the inclusion of all of these potential confounders and effect modifiers. The estimated odds ratios for being in the top 90th percentile of the distribution of these blood markers for the entire NHANES population associated with an interquartile change in PM10 (26 $\mu g/m^3$) were 1.77 (95% CI = 1.26-2.49) for fibrinogen, 1.27 (95% CI = 0.97-1.67) for platelet counts, and 1.64 (95% CI = 1.17-2.30) for WBC counts.

Seaton et al. (1999) obtained monthly blood samples from 112 elderly individuals in two cities in the United Kingdom, and investigated relationships between several blood constituents and 3-day PM10 concentrations (including modeled personal exposure and central city real-time measurements). While there was no relationship between personal PM exposure and fibrinogen, the investigators found an unanticipated pattern of PM-associated changes in blood components suggesting a sequestration of red blood cells, specifically decreased levels of hemoglobin, RBCs, and packed cell volume. In addition, there was a significant decrease in platelets in relation to personal PM exposure and a decrease in fibrinogen associated with central-city PM measurements (both of these blood components are involved in the formation of blood clots). Finally, they observed a significant increase in CRP, consistent with the recent Peters (2001b) study discussed above. Seaton et al. (1999) speculated that these results might be explained by particle-associated effects on RBC adhesive properties, making these cells more likely to be involved in thrombus formation in the circulation. The findings related to decreased RBCs, hemoglobin, platelets and fibrinogen are not entirely consistent with the results of the controlled exposure study by Ghio et al. (2000) or the cross-sectional data from Schwartz (2001b), discussed above.

If indeed PM pollution might be causally linked with increased formation of blood clots, one might also expect to see a relationship to the incidence of myocardial infarctions. One mechanism by which myocardial infarction may develop is through disruption of an atherosclerotic plaque in one of the coronary arteries; the extent to which this becomes a site for thrombus formation depends in part on the balance of forces affecting blood coagulation in the individual's circulation. Recently, Peters et al. (2001a) examined potential associations between PM concentrations and the timing of onset of symptoms in 772 patients with myocardial infarction in the greater Boston area. They found significant associations between symptom onset and both acute (within 2 hr prior to symptom onset) and subacute (24-average PM2.5 in the previous day) exposures, after adjusting for season, weather, and day of the week. Moreover, they found increasing risks with increasing PM2.5 concentrations. Adjusted odds ratios for increases in PM2.5 from the 5th to the 95th percentiles in 2-hr (25 $\mu g/m^3$, representing the range of the 2-hr average PM2.5 distribution between the 5th and 95th percentiles) and 24-hr (20 $\mu g/m^3$, representing the range of the 24-hr average PM2.5 distribution between the 5th and 95th percentiles) exposures were 1.48 (95% CI=1.09-2.02) and 1.62 (95% CI=1.13-2.34), respectively. For PM10 the comparable odds ratios for 2-hr (40 $\mu g/m^3$) and 24-hr (30 $\mu g/m^3$) averaging times were 1.51 (95% CI=1.06-2.15) and 1.66 (95%

CI=1.11-2.49), respectively. In this study the mean levels of 2-hr and 24-hr average PM2.5 were both 12.1, and for PM10 the corresponding mean values were both 19.4, though in both instances the shorter averaging intervals showed greater variability. Interestingly, the entire range of 24-hr PM2.5 concentrations in this study was lower than the U.S. EPA's ambient air quality standard for fine particles of 65 $\mu g/m^3$.

7.8.4 Disturbances of the Cardiac Autonomic Nervous System

PM-associated mortality may be explained, at least in part, by alterations in autonomic nervous system balance. Heart rate variability (HRV – a measure of the heart's ability to respond to stress), resting heart rate, blood pressure, and cardiac arrhythmias are all intimately connected with the balance between the two principal components of the autonomic nervous system – i.e., sympathetic and parasympathetic nervous systems. Numerous studies have demonstrated an association between cardiac autonomic balance and all-cause mortality (Tsuji et al., 1994), sudden cardiac death (Algra et al., 1993), and death due to congestive heart failure (Szabó et al., 1997).

HRV refers to oscillations both in the intervals between consecutive heart-beats and in consecutive instantaneous heart rates as observed on an electrocardiogram. Reduced HRV is considered a good predictor of increased risk of cardiovascular morbidity and mortality (Wolf et al. 1977; Tsugi et al. 1994; 1996; Nolan et al. 2000). HRV can be used to stratify the risk of sudden death following myocardial infarction (Kleiger et al., 1987; Copie, 1996) and in congestive heart failure (Szabó et al., 1997). A marked decrease in HRV is observed immediately preceding EKG changes precipitating ischemic sudden death; fatal arrhythmias may be triggered by such sudden autonomic dysfunction (Corbalan et al., 1974; Pozzati et al., 1996). Although decreased HRV clearly indicates a worse prognosis for individuals with heart disease, it is unknown whether this relationship is causal or whether decreased HRV represents only an epiphenomenon of more fundamental pathophysiological changes. Moreover, though several studies (described in the following paragraphs) demonstrate associations between PM exposure and HRV, the mechanistic linkage (if any) between these phenomena is unknown.

Several recent publications have linked exposure to ambient PM with decreased HRV (Liao et al 1999; Gold et al. 2000; Pope et al. 1999). There are at least a half dozen ways of measuring changes in HRV discussed in these papers, and there are some differences in results between studies. However, they are all consistent in demonstrating an inverse relationship between particulate air pollution and at least one measure of HRV. Of particular interest in these studies is the observation that these HRV changes could be observed shortly after exposure to PM (i.e., within hours).

The first published study examining the relationship between air quality and heart rate variability involved seven individuals with heart disease (congestive heart failure, angina, history of myocardial infarction, coronary artery bypass graft surgery, and arrhythmias), whose heart rates and rhythms were monitored on several occasions with and without elevated levels of particulate air pollution (Pope et al. 1999). In this small study, PM10 was associated with decreased measures of total HRV (SDNN) and long-term HRV (SDANN), but an increase in one of the short-term measures of parasympathetic tone (r-MSSD). While parasympathetic tone is generally considered to have a beneficial or protective effect, there is at least one study suggesting that increases in parasympathetic stimulation of the heart may be linked to serious arrhythmias (Kasanuki et al. 1997).

Liao and colleagues (1999) undertook standardized cardiac monitoring in 26 elderly residents of a retirement home in Baltimore over a three-week period, examining changes in HRV in relation to several concurrently measured indoor and outdoor particulate metrics. Among the

18 subjects with pre-existing cardiovascular disease, the investigators reported statistically significant, decreased HRV in relation to several indoor and outdoor measures of PM2.5 measured the same day or one day previously. Minimal, nonsignificant effects were observed among the subjects with no documented cardiovascular disease, though the number of individuals in this group was small (n=8). One aspect of the analysis included dividing each individual's HRV (specifically, the high-frequency power, an indicator of parasympathetic tone) into tertiles, and evaluating the relationships between PM2.5 levels and the position of the high-frequency power on any given day within that individual's distribution for the whole study. The investigators reported that, when the 24-hour PM2.5 concentration exceeded 15 $\mu g/m^3$, the risk of having an individual's HRV in the lowest third of his or her HRV distribution increased by three-fold, compared to days when the PM2.5 concentration was lower (OR = 3.08, 95% C.I. = 1.43 – 6.59). The clinical significance of this report is unclear; however, as cardiac parasympathetic activity is generally considered beneficial, acute decreases in this index of HRV may indicate an increased risk of an adverse cardiac event.

Gold and colleagues (2000) conducted 163 brief (25 minutes) electrocardiographic measurements in 21 ambulatory Boston residents (aged 53 to 87), once a week over a three-month period. Ambient PM2.5 and PM10 were measured in real-time with TEOMs located about 6 km from the study site. They reported a variety of statistically significant effects on two measures of HRV related to PM2.5, measured during the hour of EKG monitoring and during the three hours prior to such monitoring. No associations between PM2.5 and HRV were seen at a lag period longer than 24 hours, nor was any association noted for coarse particles. Although different metrics were used in this study than in the Liao (1999) investigation, these investigators also found a relationship between PM2.5 and decreased parasympathetic cardiac activity for a short interval preceding the measurement of HRV.

Another recent publication by Pope et al. (2001) reinforces the observations that changes in HRV can occur quite rapidly after exposure to air pollution. Sixteen volunteers were monitored electrocardiographically over the course of a day when they spent alternating 2-hour periods outside and inside a smoking lounge at a major airport. Several measures of HRV were significantly decreased in relation to several measures of exposure during the 2-hr periods in the smoking lounges. In contrast to the Liao et al. (2000) and Gold et al. (2000) reports, the measures reflecting parasympathetic tone appeared to be less strongly affected than the other measures relative to measured particles. While cigarette smoke contributes little to ambient air pollution, the rapidity of the changes observed in HRV is consistent with the findings of the studies discussed above.

Exposure to particulate air pollution has also been associated with another potentially adverse disturbance of the cardiac autonomic nervous system, as manifested by increased heart rate. Increased resting heart rate is considered an independent risk factor for cardiovascular mortality (Goldberg et al. 1996, Mensink et al. 1997). This phenomenon has not been extensively investigated in epidemiological studies. Pope et al. (1999b) found that, among 90 elderly but healthy individuals in Utah, PM10 levels were related to small, but significantly increased resting heart rates. For instance, a 100 $\mu g/m^3$ increase in PM10 (same-day) was associated with about a 50% increased risk of having at least a 10-beats/min elevation in heart rate or pulse (OR =1.51, 95% C.I. = 1.00-2.29), while PM10 lagged by one day was associated with a near-doubling of the risk of the pulse increasing by at least 10 beats/min (OR =1.95, 95% C.I. = 1.35-2.82).

In another analysis of the German participants in the MONICA study (discussed above), Peters et al. (1999) assessed whether resting heart rates increased in relation to air pollution among a subset of 2,681 men and women who had valid electrocardiographic tracings during both the 1984-85 and 1987-88 parts of the study. During the 1985 episode, resting heart rates

were increased, more so in women than in men, relative to non-episode days of the study. In addition, mean heart rates were slightly, but significantly, elevated in relation to same-day and five-day averages of TSP, sulfur dioxide, and carbon monoxide. Even excluding the episode days from the analyses, both TSP and sulfur dioxide were still both related to small, but significant changes in mean heart rates (between 1 and 2 beats/min). Though the overall mean elevations in heart rate were small, they provide support for the notion that PM air pollution is associated with altered autonomic control of the heart.

In contrast to these studies, Gold et al. (2000), in a study of elderly Boston residents, found that PM2.5 levels were associated with decreased resting heart rate. However, this finding appears to be physiologically inconsistent with the finding of decreased PM-associated short-term HRV in this panel, as described above. The investigators speculated that this inconsistency may be due to autonomic dysregulation, in which both HR and HRV might decrease in concert. In any case, there is limited evidence that ambient PM is associated with changes in heart rate in humans.

Control of blood pressure is another manifestation of the influence of the autonomic nervous system, particularly the sympathetic nervous system. Elevated blood pressure (or hypertension) is the most common cardiovascular condition in the U.S., affecting over 60 million Americans (Oparil, 1992). Hypertension is a well recognized risk factor for cardiovascular disease, stroke, and renal disease. In an examination of a subset of 2,607 participants in the German MONICA study (discussed above), 5-day average TSP (70 $\mu g/m^3$) was associated with a 1.96 mm Hg increase in systolic blood pressure (SBP), adjusting for relevant confounders and effect modifiers, including temperature, barometric pressure, and individual cardiovascular risk factors (Ibald-Mulli et al., 2001). Although sulfur dioxide was also associated with increased SBP, inclusion of both pollutants in the same regression models indicated that the TSP effect dominated that of sulfur dioxide. Interestingly, the effects on SBP were magnified in individuals with other cardiovascular risk factors: for subjects with high levels of plasma viscosity, a 90 $\mu g/m^3$ same-day increase in TSP was associated with a 6.93 mm Hg increase in SBP (95% CI = 4.31-9.75); while among those with higher resting heart rates (>90^{th} percentile, or > 80 beats/min), the same increment in TSP was associated with a 7.76 mm Hg increase in SBP (95% CI = 5.70-9.82). These findings suggest that there may be persons with pre-existing cardiovascular disease who are especially susceptible to autonomic effects of exposure to ambient particles. How PM may affect SBP is unknown, but may be related to increased blood levels of endothelin-1, a protein involved with regulating vascular tone, which has been detected in the blood of experimental animals exposed by inhalation of very high levels (40 mg/m^3) of resuspended urban particles, even though these failed to produce obvious structural pathology in the animals' lungs (Bouthillier et al. 1998). Endothelin-1 is produced not only by lung capillary (endothelial) cells, but also by airway epithelial and neuroendocrine cells, as well as macrophages. A variety of potentially adverse cardiovascular effects have been associated with elevated levels of endothelin-1, including increased blood coagulability, worsening of congestive heart failure, and increased risk of mortality after myocardial infarction (Bouthillier et al. 1998).

Finally, the incidence of serious cardiac arrhythmias has been linked with exposure to PM2.5. Implanted cardioverter defibrillators (ICDs) can initiate pacemaker activity if required, or provide an electric shock to the heart in order to terminate potentially fatal arrhythmias (ventricular fibrillation or ventricular tachycardia). An ICD logs each such event electronically. Peters and colleagues (2000b) recorded the ICD data for 100 individuals for approximately 3 years, and compared the ICD events with air pollution over this period. Overall, NO_2 and CO appeared to provide the strongest associations with ICD discharges. In the most susceptible members of this population (i.e., those with 10 or more discharges [n = 6]), however, PM2.5

and PM10 were both associated with an increased risk of an ICD discharge (OR = 1.64, 95% CI = 1.03 – 2.62; and OR = 1.68, 95% CI = 0.98 – 2.86, respectively, with a 2-day lag for each). Though the effects for both PM2.5 and NO_2 were essentially linear, including both pollutants in the same regression model reduced the PM effect to zero, while the NO_2 estimate remained unchanged. Although this study is limited by the small number of patients at high risk, and by the lack of individual clinical data other than the ICD discharges, it does suggest another potential effect of PM (as well as gaseous pollutants) on cardiac autonomic balance. A recent mortality time-series study conducted in the Netherlands (Hoek et al. 2001) provides some consistency with these findings, with risks of mortality from arrhythmia in relation to 7-day means of black smoke (40 $\mu g/m^3$, RR= 1.071, 95% CI=1.001-1.146) and PM10 (80 $\mu g/m^3$, RR=1.041, 95% CI = 0.932-1.163).

Recent publications involving PM exposures of "sick" or compromised experimental animals provide evidence supportive of these findings in humans. The compromised animal models examined in these studies include monocrotaline (MCT) treated rats, which serve as a model for emphysema, rodents with chronic bronchitis induced by high-level sulfur dioxide exposure, spontaneously hypertensive rats, and aged rodent models. Effects observed under these exposure conditions include a variety of cardiac arrhythmias, bradycardia (slowing of the heart rate), increases in plasma fibrinogen (a protein integral to blood clotting discussed above), hypertension, increases in pulmonary inflammation and mortality (Costa and Dreher, 1997; Kodavanti, et al., 1999, Watkinson, et al., 1998 and 2000; Campen, et al., 2000; Gardner, et al., 2000).

A series of experiments in spontaneously hypertensive (SH) rats is illustrative of the utility of compromised animal models. The pathophysiology of hypertension in the SH rats is similar to that observed in essential hypertension in humans. Kodavanti et al. (2000) examined normotensive and spontaneously hypertensive rats, exposed to filtered air or to high-dose (15 mg/m^3) residual oil fly ash (ROFA – a source containing high levels of the soluble metals iron, vanadium, and nickel) particles by nose-only inhalation for six hours/day for three days. They found that, compared to normotensive rats, the SH rats had evidence of pulmonary inflammation, alveolar hemorrhage, cardiomyopathy, and evidence of ST-segment depression by electrocardiography (ECG), an indicator of insufficient oxygen delivery to the heart muscle. After ROFA exposures, the SH rats showed significantly greater pulmonary injury and inflammation, including alveolar hemorrhage, a compromised ability to increase anti-oxidant defensive responses, and exaggerated depression of the ST segment on ECG (Kodavanti et al. 2000). In addition, both strains of rats exhibited similar adverse reactions to ROFA exposure, including increased airway reactivity, focal lesions in alveoli and airways, as well as around airways and blood vessels of the lung, pulmonary inflammation and production of inflammatory cytokines. Thus, although the dose levels were extremely high compared to ambient particles, this experiment suggests that compromised animals are potentially more vulnerable to pollutant-associated oxidative stress and pulmonary vascular leakage than healthy animals. Generally similar results were obtained with an experiment using one-time intratracheal administration of high-dose ROFA or nickel, but not vanadium (Kodavanti et al. 2001).

Several toxicological studies report cardiac arrhythmias in compromised animals exposed to high-dose ROFA. Investigators exposed Sprague-Dawley rats (one group with pulmonary inflammation and hypertension from MCT pre-treatment and one control group) intratracheally to large doses of ROFA (0.25, 1.0, and 2.5 mg) and observed a variety of cardiac arrhythmias in both groups (Watkinson 1998; Campen et al. 2000). However, the compromised group had more severe arrhythmias, including patterns indicative of inadequate cardiac oxygenation (myocardial ischemia) and conduction abnormalities (2^{nd} degree heart block), accompanied

by substantial mortality rate in all exposure levels (about half of the compromised animals died). In a study of rats exposed intratracheally to several different kinds of particles (ROFA, volcanic ash, and resuspended ambient particles from Ottawa, Canada), ROFA induced significant pulmonary inflammation, bradycardia and arrhythmias in healthy rats, which were exaggerated in MCT-treated rats. MCT and SH rats exposed by inhalation showed similar, but less severe, effects. Older SH rats exposed to high dose ambient particles (2.5 mg intratracheally) also exhibited significant bradycardia and cardiac arrhythmias. The volcanic dust administration had no cardiac effects in any animal group (Watkinson et al. 2000).

Rats exposed to concentrated ambient particles (whose composition can vary from day to day) were found to exhibit various degrees of pulmonary inflammation (Kodavanti et al. 2000b). In these whole-body inhalation studies, involving exposure concentrations of 475 – 907 $\mu g/m^3$, the pulmonary responses, when they occurred, were generally modest, and the animals with chronic bronchitis fared slightly worse than the control animals. Thus, although these exposure conditions were found to cause injury and inflammation, the results were inconsistent, which may have been due in part to the relatively low metal content of these particles (collected in Research Triangle Park, NC, a nonurban area). These results suggest that the very high-dose intratracheal experiments using toxic ROFA particles, for instance, may have limited generalizability to environmental exposures.

In a similar vein, Gardner et al. (2000) found increased blood fibrinogen levels in rats exposed only to the highest dose of ROFA particles by intratracheal instillation (8.3 mg/kg), but not at lower concentrations (1.7 and 0.3 mg/kg). Recognizing the limited statistical power of this investigation (six rats per exposure group), these results suggest that although animal models may help illuminate potential toxicological mechanisms, the necessity of using extremely high-dose exposures warrants a cautious interpretation.

Thus, animal studies using high-dose exposures by intratracheal administration and inhalation provide ancillary support for observations of pulmonary inflammation and cardiopulmonary toxicity in epidemiological and controlled human exposures. Such investigations bolster the biological plausibility of the human studies, but are nevertheless limited by uncertainties related to cross-species extrapolation and high-level exposures used.

7.8.5 Summary

In summary, recent research provides mechanistic support for a causal relationship between ambient PM and the cardiopulmonary morbidity and mortality consistently observed in time-series studies. Such support derives from clinical, epidemiological, and toxicological studies of a variety of pathophysiological events that could result in adverse cardiovascular outcomes. Localized airway inflammation and absorption of particles not only into the lung interstitium, but into the circulation, may result in systemic impacts, including effects on factors influencing blood coagulation, altered cardiac autonomic control, and recruitment of inflammatory cells from the bone marrow. Interestingly, most if not all of these events have been reported to occur acutely (within a day or less of exposure), and at least in the German MONICA study, several were observed to occur in concert in a subgroup of potentially vulnerable individuals. While the evidence is still fragmentary, it represents a dramatic advance from a few years ago, and begins to sketch a framework of biological plausibility for the time-series studies.

7.9 Causal Inference

This section deals with the evidence that the associations between both acute and chronic exposures to ambient PM and human morbidity and mortality represent causal relationships. The following criteria for causal inference are considered: (1) the consistency of the findings;

(2) the coherence of the study results; (3) the likelihood that findings are due to chance; (4) the possibility that findings are due to bias or confounding; (5) temporal sequence of the associations; (6) the specificity of the findings; (7) evidence for exposure-response relationships; (8) strength of the associations; and (9) the biological plausibility of a causal associations. These are based on informal guidelines for causal inference described by Sir Austin Bradford Hill, as modified by other epidemiologists (Hill, 1965; Rothman, 1988).

7.9.1 Consistency of Results Among Different Studies

The consistency of results among scores of epidemiological studies provides substantial evidentiary support for causality. Several hundred studies, conducted among different populations on five continents over multiple time periods, have reported small, but consistently elevated risks of daily mortality and diverse measures of morbidity (such as hospital admissions and emergency department visits for cardiac and respiratory causes, exacerbation of asthma, increased respiratory symptoms, restricted activity days, school absenteeism, and decreased lung function). Though the principal study design has been time-series analysis, modeling approaches have differed substantially among investigators; moreover, similar estimates of effect have been obtained with other study designs, including case-crossover and panel studies. The ranges of risk estimated in all these studies have been remarkably similar, despite the different PM source mixtures and size distributions, co-pollutant distributions, weather patterns, population characteristics (distributions of age, baseline health status, and access to health care) (See Section 7.3, for example). Daily mortality and morbidity have also been linked with different measures of PM, as well, including TSP, PM10, PM2.5, the coarse fraction (PM10-PM2.5), black smoke, and ultrafine particles. In general, consistency of results across scores of investigations offers one of the strongest arguments favoring a causal relationship (Ostro, 1993).

7.9.2 Coherence of Results

Referring in particular to the time-series studies of mortality, Bates (1992) has argued that, if the PM-mortality relationship is causal, there should also be evidence of relationships between PM and health outcomes of lesser severity, such as hospitalizations, changes in lung function, and so forth, suggesting an ensemble of coherence among possible outcomes. This phenomenon has been observed in a number of areas throughout the world; perhaps the best illustration of such coherence in a given area are the studies undertaken in the Utah Valley. In addition to increases in PM-associated mortality, studies in this area have demonstrated statistically significant relationships between ambient PM and respiratory hospitalizations, decrements in children's lung function, school absenteeism, respiratory symptoms, medication use among asthmatics, increased heart rate and decreased heart rate variability among elderly individuals (Pope, 1996, 1999a, 1999b).

7.9.3 Likelihood That the Findings are Due to Chance

Almost all the studies described in the previous sections showed increased risks of PM-associated morbidity and mortality, though these results are not all statistically significant. The purpose of significance testing is to compare the results of a given study with what would be expected to occur by chance if the null hypothesis of no effect (e.g., between ambient PM exposure and daily mortality) were true. This assessment is usually based on comparison with a pre-designated significance level (usually 5%), which indicates a rough cut-off value for assessing the likelihood of the results that could be expected to occur by chance. Thus, finding that the results are statistically significant is a judgment that the results are not likely to be due to chance. Moreover, it should be noted that many of the results cited above are highly statistically significant, indicating that they are extremely unlikely to be due to chance. Also, it can be seen in Table 7.1 and Sections 7.3 through 7.6 that, with few exceptions, there

is a consistent tendency for point estimates of relative risk to be greater than unity. If these findings were due to chance, one would expect a more nearly equal distribution of point estimates of risk above and below unity.

7.9.4 The Possibility That Findings are Due to Bias or Confounding

In evaluating these results, one needs to consider confounding, information bias and selection bias. In the time-series studies that are population-based, selection bias is not an important issue. Rather the principal concerns regarding the validity of the results would be confounding and information bias, specifically the potential impact of misclassification of exposure.

Confounding occurs when the estimates of effect are distorted by an extraneous variable that is associated with both the exposure and outcome of interest, where that extraneous variable is not part of the causal pathway between the exposure and the outcome. In daily time-series analyses, any confounder would have to vary in concert with both the daily fluctuations in pollutant concentrations and with the health outcome. Thus, variables that one might intuitively consider as potential confounders, such as cigarette smoking patterns, are not relevant in this context. The principal potential confounders of concern in such studies are meteorological variables and gaseous co-pollutants such as ozone and sulfur dioxide, and possibly the presence of respiratory epidemics such as influenza.

Of the meteorological variables, temperature is probably the most important, as it has been demonstrated to have independent effects on a variety of health outcomes, including mortality. All of the time-series studies of PM and mortality cited in this report have controlled for temperature, or have at least examined whether temperature could be a confounder. Investigators have employed a variety of modeling approaches to assess the impact of temperature; some studies have undertaken sensitivity analyses to assess the likelihood that weather-related impacts were being inappropriately ascribed to PM (Samet et al., 1998; Pope and Kalkstein, 1996). The weight of the evidence indicates that the PM-associated health outcomes are not the result of confounding by temperature or other meteorological variables. In addition, similar estimates of PM-related effects have been obtained in cities with diverse climates and different seasonal relationships between PM and temperature. This issue is discussed in greater detail in Section 7.3.

Respiratory epidemics, such as influenza, regularly occur in specific seasons (e.g., influenza generally is a winter phenomenon in the United States). To the extent that there is adequate control of seasonal meteorological influences in any given study, this should address potential effects of confounding by infectious disease. In addition, if PM-associated mortality or morbidity is also observed in other seasons in a given locale, this would indicate that respiratory infectious disease epidemics could not explain the association. In some instances it would be methodologically inappropriate to control for influenza, for example, if this outcome itself represents either one of the health outcomes of interest or can be considered part of the causal pathway for one of the health outcomes, such as exacerbation of asthma. Several studies have explicitly modeled infectious respiratory illness outbreaks in examining PM-associated health effects; these also indicated that the relationships could not be explained by seasonally concurrent epidemics (Braga et al., 2000).

Finally, there is the issue of confounding by gaseous co-pollutants, including specifically ozone, nitrogen dioxide, sulfur dioxide, and carbon monoxide. All of these pollutants have also been associated in time-series studies with daily mortality and a variety of other adverse health outcomes. Therefore, in the presence of strong correlations between any one or more gaseous pollutants with a PM metric within a given study, it may be difficult to disentangle their relative impacts. In some instances, particularly in studies outside of North America, measurements of co-pollutants were limited, and therefore the potential impacts of these

gaseous pollutants could not be controlled for in the analysis. The two principal methods to address potential confounding by gaseous pollutants are: (1) to examine PM effects in multiple locations in which there are different correlations between PM and the various gases; and (2) to include multiple (measured) pollutants in the regression model. Using the first method, if the PM coefficients are consistent from place to place in the presence or absence of a putative co-pollutant confounder, this suggests that the associations between PM and mortality or morbidity indices are independent of, and not confounded by, the other pollutants. In view of the plethora of epidemiological studies in diverse locations, some with high ozone or sulfur dioxide levels, and some with low concentrations of these pollutants, the evidence is compelling that PM effects cannot be explained away due to confounding by co-pollutants.

In a recent, large-scale application of the second method involving 90 U.S. cities, Samet et al. (2000) sequentially tested the estimated effects of PM10 on daily mortality after each of the principal gaseous pollutants (ozone, nitrogen dioxide, sulfur dioxide, and carbon monoxide) was added to the regression model. These authors reported trivial or no change in the estimated PM10 coefficients when the other pollutants were included in the model. Similar results have been obtained in most of the studies that have examined PM10 and mortality, with few exceptions (e.g., Moolgavkar, 2000). Other recent examinations of the problem of confounding by co-pollutants have also found little evidence that confounding can explain the associations between PM concentrations and adverse health outcomes (Schwartz, 2000a, Katsouyanni et al., 2001).

One other potential threat to validity of the results of epidemiological studies is information bias, particularly in the form of measurement error. In this instance we are concerned with errors in measurement of PM exposures. Such measurement error is an inherent feature of epidemiological studies: given that pollutant concentrations vary over space and time, as do individuals' activity patterns, it is not possible to measure personal exposures to the important components of PM for large numbers of individuals. This is a multi-dimensional problem that could consist of the following components: (1) use of a PM metric that includes some "nuisance" particles that do not really contribute to health effects rather than the "true" components that are biologically active; (2) errors in measurement between the values recorded by ambient monitors and the true ambient levels, due to either instrument error or temporal-spatial variation, or both; (3) differences between aggregate ambient measurements and individual personal exposures; (4) differences between average personal exposures and true ambient pollutant levels; and (5) differences in the accuracy of measurement of co-pollutants, so that in multivariate regression models, those pollutants measured with greater accuracy and precision may spuriously appear to have a greater effect than they would if all were measured with equivalent accuracy and precision.

Typically the effects of measurement error tend to bias the results towards the null hypothesis of no effect – that is, the effects of PM on morbidity and mortality tend to be underestimated. There may be exceptions to this generalization, however. Recently the issues of measurement error in air pollution time-series studies were systematically reviewed, characterizing the errors in measurement as either classical or Berksonian in nature (Zeger et al., 2000). Berkson-type errors, an example of which is using aggregate rather than individual exposure data, do not produce biased regression coefficients. Zeger and colleagues (2000) suggest that in the usual case, time-series studies will tend to underestimate, rather than overestimate, pollutant effects. In the case of multi-pollutant models, differences in the monitoring accuracy and precision of pollutants may result in confounding, with the effects of a more poorly measured pollutant being transferred to one measured more accurately, but only when the pollutants or their errors in measurement (particularly the latter) are strongly correlated. When pollutant levels are strongly correlated, they generally would not be included

in the same regression model, as this produces unstable and biased estimates of effect. Zeger et al. (2000) suggest that the largest potential source of bias in measurement error is likely to be due to differences between ambient measurements and average personal exposures, which could occur if indoor sources produce particles of similar size and toxicity as outdoor local and regional sources. Taking the "best" data set available that would allow an examination of the magnitude of this kind of error (from the P-TEAM study in Riverside, CA), they found again that standard regression analysis will tend to underestimate the strength of the association between fixed-site monitoring data and adverse health outcomes (mortality, in this case).

Based on the above, it is possible that, in limited circumstances, particularly when multiple pollutants are measured with error, that some of the PM effect may be due in part to differential measurement error. However, it is reasonable to infer that in most situations, the results of the numerous time-series studies of PM-associated morbidity and mortality cannot be explained by information bias.

7.9.5 Temporality of the Associations

That a putative cause precede its effect(s) is a *sine qua non* for causal inference (Rothman, 1982). It is in this sense that this guideline for causal inference is typically used in epidemiology, and is clearly met in the ensemble of PM studies. In the time-series studies of morbidity and mortality, one typically finds significant associations between PM concentrations and adverse health outcomes with lags of zero to four days, with moving average concentrations occasionally demonstrating a slightly stronger association. Several studies examining "reverse lags" (i.e., with the health effects preceding the pollution measurements) have found no relationship.

However, a number of investigations have found statistically significant associations between PM concentrations and adverse health outcomes on the same day. For certain health outcomes, such as exacerbation of asthma, this could be explained mechanistically without much difficulty. For cardiovascular outcomes, including mortality, such short lags between exposure and outcome might appear problematic. Nonetheless, recent evidence suggests relatively rapid systemic responses to PM pollution that are consistent with the observations in the time-series studies (Gold et al., 2000, Pope et al., 2001 – see section 7.8 above).

7.9.6 Specificity of Effect

In the original formulation of the guidelines for causal inference, Hill (1965) expressed the notion that the basis for causal inference would be strengthened if an exposure led specifically to a single effect. The absence of such specificity does not necessarily negate the existence of a causal relationship – witness the protean manifestations of disease engendered by exposure to cigarette smoke. Nevertheless, it is intuitive that the more specific an association between an exposure and an adverse health outcome, the more likely it is to represent a causal relationship. Although PM exposures have been linked to a variety of adverse effects, the latter are circumscribed to effects on the respiratory and cardiovascular systems. Given our current understanding of the pathophysiology of inflammation, with both local (respiratory) and systemic effects, these are the organ systems that one would expect to be most strongly affected by exposure to particles. While many of the mortality time-series studies have examined impacts on total mortality only, a few have done comparative analyses of relationships with cardiac- and respiratory mortality and with mortality from all other causes (Section 7.3). The results of these studies suggest that the relationship between PM exposure and mortality is relatively specific to those organ systems expected to be affected by such exposures. A similar pattern can be observed with time-series studies of hospitalizations (Section 7.5).

7.9.7 Evidence for Exposure-response Relationships

As noted above, the data from most of the time-series studies discussed in this document clearly demonstrate statistically significant exposure-response relationships. The range of the PM-mortality coefficients is surprisingly narrow over a wide range of PM concentrations over time and across locations, indicating that, at least within the observable range in most metropolitan areas examined, this relationship is more or less linear (Section 7.3). Generally, for morbidity outcomes that are more common than daily deaths, the magnitude of the associations are slightly greater, as one would expect if the relationships were causal.

7.9.8 Strength of Association

The relative risk (RR) estimates obtained in the epidemiological studies of morbidity and mortality are generally low, with virtually all estimates of effect less than two. RR estimates of this magnitude may weaken the evidence of causality, due to the possibility of uncontrolled confounding or other sources of bias producing the findings. However, small estimates of relative risk do not, in themselves, nullify the existence of a causal relationship. As indicated below, the potential threats to the validity of any given study (i.e., bias and confounding) are not likely explanations of the consistent findings of increased PM-associated risks of morbidity and mortality.

In addition, when either the outcome measures or the exposure metric are given greater precision, the estimate of effect increases, which is what one would expect if the relationship is causal. For instance, as indicated in Section 7.3 and Tables 7.1 and 7.2, the risks of mortality associated with PM10 range from 0.5% to 1.6% per 10 $\mu g/m^3$ of PM10, while the likely range for PM2.5 is 1% to 2.5% per 10 $\mu g/m^3$. For cardiac and respiratory causes of death, the corresponding ranges per 10 $\mu g/m^3$ increase in PM10 concentration are 0.8% to 1.8% and 1.3% to 3.7%, respectively (Ostro et al., 1999a). This was highlighted in a recent publication from the Netherlands, in which specific causes of cardiorespiratory mortality were found to be more strongly related to PM10, than mortality overall (Hoek et al., 2001).

Thus, although the estimates of effect are low, they are consistently highly statistically significant, and increase in magnitude and precision with better specification of either the outcome or the exposure metric.

7.9.9 Biological Plausibility of the Associations

Biological plausibility is not necessary for causal inference from epidemiological studies, since it depends on the state of knowledge of ancillary disciplines. When present, however, supporting evidence from other scientific fields such as toxicology can strengthen the case for a causal association between an exposure and a disease outcome. A decade ago biological plausibility for a causal linkage of ambient PM with mortality or with multiple indicators of morbidity would have been purely speculative. Major recent advances in toxicology, clinical exposure studies, and epidemiological studies with intermediate endpoints all suggest that most of the effects observed in the epidemiological studies are likely to be initiated with inflammatory responses in the lung, which can have both local and systemic effects. While the picture is far from complete, plausible biological mechanisms of effect have been proposed and are the subject matter of active research. This issue is discussed in much greater detail in Section 7.8.

7.9.10 Summary

The scientific evidence linking PM exposure to premature mortality and a range of morbidity outcomes appears to meet the generally accepted guidelines for causal inference in

epidemiology (Hill 1965). Much current research is now focusing on biological mechanisms in order to provide a more complete understanding of the effects of PM.

7.10Benefits Assessment of the Health Effects of Particulate Matter

7.10.1 Background

The objective of this health benefits assessment is to generate quantitative estimates associated with current particulate matter concentrations in ambient air in California, and to indicate the potential benefits associated with controlling PM. In so doing, this assessment also provides information about the consequences of alternative standards for PM. Together with other information generated in this report, this assessment assists in selecting a reasonable range for PM standards that will protect public health with an adequate margin of safety. Given the lack of a discernible threshold for health effects, no PM ambient air quality standard above background levels can be risk-free. Therefore, the adequacy of the margin of safety in this case is based on a judgement that balances the nature and level of risk and the degree of uncertainty in the health effects estimates.

The evidence of significant health effects associated with PM is remarkably consistent, and meets the generally accepted guidelines for causal inference (See section 7.9). Nevertheless, the quantitative risks developed in this document include significant uncertainty. These uncertainties exist in both the development of the appropriate concentration-response functions, relating alternative concentrations of PM to changes in risk, and in the estimates of population exposure. There is also uncertainty to the extent that PM10 and PM2.5 are used as indicators for the geographically and temporally heterogeneous mixtures of pollutants in ambient air. Therefore, the risk estimates developed in this section should not be construed as precise predictors of risk. We attempt to indicate the uncertainty both quantitatively, by applying confidence bounds around the estimates, and qualitatively, in our discussion of the results.

There have been several recent published efforts to estimate the health benefits associated with reducing population exposures to PM. Ostro and Chestnut (1998) generated estimates of the health benefits associated with U.S. EPA's proposed standards for PM2.5. Kunzli et al. (2000) estimated the health effects attributed to traffic-related PM in three European countries. The U.S. EPA has embarked on several significant efforts to quantitatively evaluate the health risks associated with exposure to ambient PM10 and PM2.5. For example, the Staff Paper for particulate matter (U.S. EPA, 1996) summarized an analysis of health risks associated with attainment of alternative standards for PM2.5 and PM10. Section 812 of the federal Clean Air Act required the U.S. EPA to conduct an analysis of the health benefits of current federal air pollution legislation, which resulted in a report to the U.S. Congress (U.S. EPA, 1999). Also, EPA recently issued a Regulatory Impact Analysis required under Executive Order #12291 on the health benefits associated with new standards for heavy duty engine/diesel fuel (U.S. EPA, 2000). These efforts have undergone years of public review and comment as well as full peer review by the U.S. EPA's independent Science Advisory Board. We have, therefore, drawn considerably from prior efforts at the federal level, particularly in the development of concentration-response functions. Most of the functions used in this section are similar to those adopted by the U.S. EPA (2000). We have also added California-specific concentration-response functions, whenever possible.

To estimate the health effects (and potential benefits of PM reduction) associated with current ambient PM, four components are needed: (1) the quantitative relationship between ambient concentrations and the health response or "concentration-response" functions; (2) the exposed populations, (3) the baseline incidence rates of specific adverse health outcomes; and (4) the projected change in air pollution concentrations under consideration.

For most of the health endpoints estimated in this analysis, we considered changes from current ambient concentrations of PM10 and PM2.5 to our proposed annual average standards. Chapter 10 also provides estimates associated with their estimated background levels, since no threshold of effect has been identified for any PM metric. Changes from current ambient levels to alternative concentrations, including background concentrations, and additional details of the analysis are supplied in Chapter 10. Baseline incidence data have been extracted from available county-specific census data for California. Population data were derived from the latest census estimates. Finally, the concentration-response functions were developed from epidemiologic studies on mortality and morbidity, reviewed in Sections 7.3 through 7.6. These studies provide estimates of additional risk per $\mu g/m^3$ PM10 or PM2.5. The ultimate aggregate number of expected cases associated with current ambient concentrations is the product of these four factors.

7.10.2 Methodology: Exposure and Population Data and Assumptions

Concentrations of PM vary spatially depending on local and regional PM sources, climate patterns, and topography, as well as particle size, aerodynamic behavior, chemical reactivity, and other physico-chemical characteristics. Accordingly, population exposure estimates tend to be more accurate when the population data are highly resolved, such as at the census tract level. Population counts by census tract provide a convenient basis for determining population exposures to air pollutants. In addition, demographic data, such as age distributions, are available for each census tract.

For the estimates of health risks from PM, concentrations from a network of air quality monitors are used to determine appropriate values at each census tract. The concentration for a census tract is a weighted average of the concentrations at all monitors within a maximum allowed distance. For the present analyses of PM10 and PM2.5, the maximum distance was 50 kilometers (75 km in the Great Basin Valleys Air Basin). A small number of census tracts are more than 50 km from any PM monitor, so their populations were not included in the analyses. The population numbers will be affected slightly by different choices for the maximum distance. The weight assigned to each monitor is the inverse square of its distance from the census tract. In this way, nearby monitors are more influential than distant monitors. Although vertical physical “boundaries,” such as mountain ranges, were not used in the model, local monitors on either side of such boundaries will dominate the calculated concentrations for census tracts in their respective regions.

The basic procedure for determining exposures was first adopted by the ARB in 1993 to fulfill the requirements of Section 39607(f) of the Health and Safety Code. Full details are provided in *Guidance for Using Air Quality-Related Indicators in Reporting Progress in Attaining the State Ambient Air Quality Standards* (September 1993). For this application, the concentrations and populations were associated by census tract and merged to assemble a statewide distribution of exposures to different concentrations of PM. Ultimately, annual averages for PM10 and PM2.5 were calculated for each census tract and then, using population-weighted averages, aggregated to the county and then air basin level. For PM10, monitoring data for 1998 through 2000 were used for all monitors in the State meeting quality assurance criteria for valid data. Use of three years reduces the influence of a single year that might be less representative of long-term exposures. For PM2.5, only data for 1999 and 2000 were available.

Projected census tract data based on the 1990 census were used to develop the current population-weighted PM exposure levels, as the 2000 data were not yet available in census tract format. County-wide age breakdowns from the 2000 census were used to derive the

estimates of county–level, age-specific mortality and morbidity effects. Also, the census data contain the shape, size and centroid of each census tract.

7.10.3 Methodology: Concentration-response Functions

As mentioned above, many of the concentration-response functions have been adopted from U.S. EPA (2000), and their use and development has undergone extensive scientific peer review. Only studies based on actual measurements of either PM10 or PM2.5 as a measure of PM were used in order to reduce uncertainties in adjusting from other measures such as black smoke or sulfates. From the epidemiologic studies, an estimated risk per $\mu g/m^3$ was calculated for PM10 and/or PM2.5. As reviewed above, effects have been demonstrated for both fine and coarse particles, as well as for PM10. Therefore, estimated coefficients of PM2.5 or PM10 were converted from the original studies into an effect per $\mu g/m^3$ and applied to hypothetical changes for both PM10 and PM2.5. Tables 7.7 and 7.8 summarize the concentration-response functions for the subset of health outcomes reviewed in this section. Specifically, the estimated effects of PM on mortality and hospitalization are provided because of the severity of these impacts. In addition, effects on lower respiratory symptoms in children are provided as a measure of the magnitude of a less severe outcome on the younger population. The functions and results for the full range of estimated health effects are provided in Chapter 10.

7.10.3.1 Mortality

As reviewed in Sections 7.3 and 7.4, both short-term (daily or multi-day) and long-term (a year to several years) exposures to PM have been associated with mortality. Long-term exposure estimates are preferable since they include the effects of both long and short-term exposure and clearly represent a significant reduction in life expectancy. For long-term exposure, we rely on the estimates from the re-analysis by Krewski et al. (2000) of the American Cancer Society (ACS) cohort data originally analyzed by Pope et al. (1995). We used the Krewski et al. (2000) coefficients rather than those of Pope et al. (1995), since the former used the mean rather than median estimates of long-term concentrations of PM2.5. The mean may better reflect the long-term cumulative exposure to PM and is more sensitive to peak concentrations. In the U.S. EPA analysis, the relative risk and expected numbers of cases were lower when the estimates were based on the mean (Krewski, 2000; U.S. EPA, 2000).

Although Krewski et al. (2000) also provided estimates based on the Harvard Six-Cities mortality study (Dockery et al. 1993), U.S. EPA relied only on the Krewski analysis of the ACS data, which was based on the mortality experience of 295,000 subjects from 50 cities (including 8 in California), while the Harvard study examined 8,111 subjects in 6 mid-west and East Coast cities. Although the ACS database had a much larger sample size and included more locations, the exposure assessment in the study was probably less accurate than in the Harvard study, the follow-up period was only half of that of the Harvard study, and the population covered was slightly older (subject ages > 30 in the ACS cohort versus subject ages > 25 in the Harvard cohort). The Dockery et al. (1993) analysis of the Harvard data, as well as the Krewski et al. (2000) reanalysis, generated a larger effect on premature mortality than did the analyses of the ACS cohort.

Based on the estimates derived from the ACS cohort, we applied the estimate of a 4.62% ($\pm$ 1.2) increase in annual mortality per 10 $\mu g/m^3$ of PM2.5 to the population older than age 30. For PM10, this coefficient was adjusted by the PM2.5/PM10 ratio of 0.5 (a lower bound estimate of the ratio was assumed for this purpose), which assumes that only the fine particle share of PM10 is toxic. This adjustment was based on the re-analysis of the ACS data set by Pope and others cited in Krewski et al. (2000), which shows that for long-term exposure,

coarse particles were not associated with mortality. Note that the other major prospective cohort long-term exposure study did find apparent associations between PM10 and mortality, therefore this assumption leads to much lower effects from PM10.

Although the expected cases of premature mortality for infants (see section 7.7.3) could be added to this estimate, the underlying data are not as compelling as those from the ACS cohort. As discussed in Section 7.7.3, these studies involve either a cross-sectional study design with the attendant uncertainties relating to exposure assessment, or a time-series study design in areas with very high ambient pollution concentrations. Therefore, because of the uncertainties involved in extrapolating these effects to California, no specific mortality estimates from chronic exposure are provided for this subgroup. However, mortality effects on infants are included in the estimates based on the time-series analyses of all-cause mortality for the total population.

To generate a lower bound on cases of premature mortality, we used estimates from the short-term exposure studies. For the most part, many of the effects related to short-term exposure are subsumed by the long-term exposure estimates, so the estimates based on short- and long-term exposure should not be added together. Rather, estimates based on the short-term studies can be viewed as a minimum level of effect, and would not capture, for example, cases of lung cancer or heart disease that might be due to long-term PM exposures. Our estimates for PM2.5 are derived from the Schwartz (1996) study that used PM2.5 and a similar methodology to analyze mortality in six U.S. cities. The pooled analysis of these cities implies an increase of 1.43% (± 0.13%) in annual mortality per 10 $\mu g/m^3$ of PM2.5, an increase similar to other recent studies using this metric. For short-term exposure effects of PM10 on mortality, we pooled the results of the 11 studies conducted in California for which results were available (see Chapter 10 for details). This includes results from the counties of Santa Clara, Los Angeles, San Bernardino, Riverside, San Diego, Orange, and Alameda. The pooled effect estimate for these counties is 0.84% (± 0.20) change in annual mortality per 10 $\mu g/m^3$ PM10.

7.10.3.2 Hospitalization

For hospitalization for respiratory and cardiovascular diseases, we relied on the analysis of 14 cities by Samet et al. (2000). This includes the cities of Birmingham, Boulder, Canton, Chicago, Colorado Springs, Detroit, Minneapolis/St. Paul, Nashville, New Haven, Pittsburgh, Provo/Orem, Seattle, Spokane, and Youngstown. This analysis, sponsored by the Health Effects Institute, provided separate concentration-response functions for hospitalizations for chronic obstructive pulmonary diseases (bronchitis and emphysema), pneumonia, and all cardiovascular disease for people 65 and older. In addition, hospitalization for asthma is based on Sheppard et al. (1999) for people 64 and above in Seattle. An alternative estimate for hospital admissions was generated using the final report to the South Coast Air Quality Management District by Van Den Eeden et al. (1999). This study used Kaiser data on hospital admissions for all age groups in the Los Angeles basin.

7.10.3.3 Lower Respiratory Symptoms

To provide an indication of a less severe but more common effect on children, concentration-response estimates were developed for lower respiratory symptoms based on Schwartz et al. (1994). This study involved children ages 7 to 14 from the Harvard Six-City study and determined days with lower respiratory symptoms (reports of at least two among cough, chest pain, phlegm, and wheeze) associated with PM2.5. These estimates were applied to all children in California ages 7 to 14.

As indicated in Chapter 10, there are other significant risks associated with PM10 and PM2.5, including cases of acute and chronic bronchitis, asthma attacks and emergency room visits, upper respiratory symptoms, days of work loss and days with some restrictions in activity.

7.10.4 Risk Estimates

Applying results from the available epidemiologic studies to California data on PM suggests significant effects for both mortality and morbidity. For example, applying the prospective cohort, long-term exposure studies, the change in ambient PM2.5 from current levels in California (as described in Chapter 6) to an annual average of 12 $\mu g/m^3$ for all California counties is associated, in the long term with 6,500 (95% CI = 3,200 to 9,800) fewer cases of premature mortality per year, or about 2.9% of all mortality in the population above age 30. Attaining a concentration of 15 $\mu g/m^3$ PM2.5 for all California counties is associated with 4,000 (95% CI = 2,000 to 6,000) fewer cases of premature mortality per year.

Use of the short-term exposure studies, which only capture part of the total effects on mortality, generates a mean estimate of 2,600 fewer premature deaths per year (95% CI of 2,200 to 3,100) using a standard of 12 $\mu g/m^3$ PM2.5 and 1,700 (95% CI of 1,400 to 2,000) with a standard of 15 $\mu g/m^3$.

Mean annual estimates of reduced hospitalization associated with moving from current concentrations of PM2.5 to 12 $\mu g/m^3$, are 600 for COPD, 900 for pneumonia, 1,500 for cardiovascular disease and 500 for asthma. These effects are all associated with relatively short-term exposures to PM; no effects associated with long-term exposures are included in the hospital estimates. These estimates are fairly close to those derived using the California Kaiser data on hospitalization, which suggest a reduction of 2,100 cases of hospitalization for circulatory diseases, 1,500 for chronic respiratory disease and 700 for acute respiratory disease. Finally, among children ages 7 to 14, current concentrations of PM2.5 are estimated to result in about 209,000 (95%CI 81,000 – 323,000) excess days of lower respiratory symptoms per year.

The estimated health benefits associated with meeting lower annual averages of PM10 are also significant. These estimates are an alternative and not in addition to the PM2.5 estimates. Based on the analysis of Krewski et al. (2000) of the ACS cohort, long-term effects are only attributed to the fine particle share of PM10, not to all of PM10. As noted above, the other major prospective cohort long-term exposure study (Dockery et al., 1993) did find an apparent association between PM10 and mortality, therefore this assumption leads to lower effects from PM10. In addition, several morbidity endpoints appear to be associated with long-term exposure to PM10. Applying the prospective cohort, long-term exposure studies, the change in ambient PM10 from current levels in California (as estimated in Chapter 6) to an annual average of 20 $\mu g/m^3$ for all California counties is associated, in the long term, with 6,500 premature deaths (95% CI = 3,200 to 9,800), about 3% of all mortality for the cohort above age 30. Use of short-term exposure studies generates a mean estimate of 3,000 (95%CI = 1,600 to 4,400) premature deaths per year. Attaining a uniform concentration throughout the state of 25 $\mu g/m^3$ PM10, generates the mean estimate of 4,500 fewer deaths per year related to long-term exposure or about 2.1% of total mortality.

7.10.5 Uncertainties

Among the major uncertainties in the risk estimates is the degree of transferability of the concentration-response functions from different cities in the U.S. to California. However, eight California cities were included in the long-term exposure-mortality study (Krewski et al., 2000), which involved a total of 63 cities, while the short-term exposure-mortality estimates were derived from nine studies of California cities (see Chapter 10). Similar risk estimates for

mortality associated with acute PM exposure have been observed in over 60 cities throughout the world. In addition, similar quantitative estimates of the morbidity outcomes have been reported in multiple cities and/or have been conducted in California. Therefore, generalizing these results appears reasonable. There is still some uncertainty, however, concerning the choice of the specific studies and concentration-response functions used in this risk assessment. In this case, we used concentration-response functions that had been reviewed and judged as acceptable by U.S. EPA's Science Advisory Board. For example, although we used the results of single-day exposures in the short-term exposure-mortality studies, application of studies using multi-day averages would have generated higher effect estimates. As another example, the prospective cohort studies using the results from the ACS (Pope et al., 1995) and Harvard Six-Cities (Dockery et al., 1993) cohorts could have been pooled, producing a higher estimate than relying on only the Pope et al. study.

A second major uncertainty relates to the existence of a threshold. This is discussed in detail earlier, with the conclusion that there is no evidence for a threshold in the studies that have explicitly examined the issue. In addition, studies have demonstrated effects at very low concentrations of PM (see Table 7.1 and Figure 7.1, for example).

A third uncertainty involves the issue of co-pollutants. Specifically, it is possible that some of the estimated health effects include the effects of both PM and other correlated pollutants. Many of the daily exposure studies isolated an independent effect of PM and/or tested for possible interactions or joint effects with other pollutants. However, given inherent errors in measurement of exposure to ambient pollutant, it is possible that PM is serving as an index for a mix of combustion-related pollutants or other sources of pollutants. It should be noted, however, that SB25 requires OEHHA to consider possible effects of exposure to multiple pollutants in evaluating ambient air quality standards. Thus, insofar as the PM concentration-response association may include effects of other pollutants, this is in accordance with the statutory requirements. Related to this issue is the lack of a clear understanding of the relative effects of fine versus coarse particles. In addition, there is uncertainty related to the use of the existing network of monitors to represent current ambient concentrations. There will be some error in these measurements, depending on the location of these monitors and the spatial pattern of the pollutants.

Finally, estimates for only a subset of adverse outcomes are provided. For example, estimates of the effects of PM on cancer incidence and infant mortality are not provided. In addition, no estimates on averting behavior are provided. This would include measures that are taken to prevent symptoms from occurring in the first place, such as avoiding strenuous exertion on days with high PM, staying indoors, use of prophylactic medication, purchasing of air filters, and so forth.

7.10.6 Summary

In summary, the epidemiologic evidence and risk assessment support the likelihood of significant mortality and morbidity effects related to current exposure to PM. Although the relative risk per unit is low, the large number of people exposed suggests the existence of a potentially major impact on public health. A precise measure of risk, however, is difficult to determine. Given the above uncertainties, it is more likely that we have underestimated rather than overestimated the effects of PM.

7.11Recommendations for Standards

This chapter presents the staff recommendations for the Board to consider in promulgating the PM Ambient Air Quality Standards (AAQSs) for California. The section begins with findings on the overall adequacy of the current standards for PM with respect to protecting the

health of the public, including infants and children. It continues with recommendations for the pollution indicators, averaging times, forms, and concentrations adequate to protect public health.

The recommended concentrations for the PM standards should be based on scientific information about the health risks associated with PM, recognizing the uncertainties in these data. With this in mind, the numerous studies of PM-associated morbidity and mortality indicate that, within the concentration ranges reported, there is no identifiable "bright line" or threshold PM concentration for either short- or long-term exposures, below which health effects would not occur. However, the Children's Environmental Health Protection Act [Senate Bill 25, 1998 Legislative Session, Escutia; specifically California Health & Safety Code Section 39606(d)(2)] does not require setting a given AAQS at a level that ensures zero risk. Given the current state of the science, it would not be possible to set such standards for particulate matter. Rather, the statute requires a standard that "adequately protects the health of the public, including infants and children, with an adequate margin of safety."

The governing statutory language indicates that California's ambient air quality standards should also protect other vulnerable populations, in addition to infants and children, and the general public [(H&SC sections 39606(d)(2) and 39606(d)(3)]. This legislative directive is consistent with historical practice in California, where ambient air quality standards have been formulated to protect identifiable susceptible subgroups, as well as the general population. For instance, the one-hour sulfur dioxide standard was developed in order to protect the most sensitive recognized subgroup, exercising asthmatics. Nonetheless, even with standards tailored to shield vulnerable populations, there may be exquisitely sensitive individuals remaining outside the ambit of protection.

Although both the California Health & Safety Code (section 39606) and the federal Clean Air Act (section 109) refer to an adequate margin of safety, no specific legislative definition of "adequate" is provided. This judgment is left to the responsible regulatory agencies. As described in the preceding chapters, the current epidemiological data suggest linear relationships between adverse health outcomes and ambient PM concentrations, with no clear demarcation of a level of PM exposure below which no adverse health effects would ever be expected to occur. The incorporation of a safety margin has been recognized by the California Supreme Court as integral to the process of promulgating ambient air quality standards [Western Oil and Gas Association v. Air Resources Board, 22 ERC 1178, 1184 (1984)]. To the extent that health effects associated with ambient PM have occurred at relatively low levels of exposure, and that there is substantial inter-individual variability in response to environmental insults, it is unlikely that any PM standard will provide universal protection for every individual against all possible PM-related effects. Thus, in this instance, applying the notion of an "adequate margin of safety" for PM standards becomes somewhat challenging. Nevertheless, taking into account the limitations of the scientific data, we have operationalized the concept of an adequate margin of safety by recommending standards that, when attained, should protect nearly all of the California population, including infants and children, against PM-associated effects throughout the year.

7.11.1 Adequacy of Current California AAQS for PM in Protecting Public Health

The extensive epidemiologic data on the health effects of PM, supported by clinical and toxicological evidence, suggests that in combination the current annual average standard for PM10 of 30 $\mu g/m^3$ and the 24-hour average of 50 $\mu g/m^3$ do not offer sufficient protection of public health, including that of infants and children (ARB, 2000). Chronic exposures to ambient PM appear to be especially deleterious, and may influence responses to shorter-term (usually daily) exposures. As reviewed in the above sections, there are strong and consistent

associations between daily exposure to PM (measured as PM10, PM10-PM2.5, or PM2.5) and a range of adverse outcomes, including premature mortality, hospital admissions, emergency room and urgent care visits, asthma exacerbation, chronic and acute bronchitis, restrictions in activity, school absenteeism, respiratory symptoms, and reductions in lung function. These studies have been conducted in a wide range of cities on five continents, with differing PM sources, climates, seasonal patterns, co-pollutants, and population characteristics. The more severe outcomes are experienced primarily by the elderly and by people with pre-existing chronic heart or lung disease. However, several epidemiological studies suggest that children under age five may also experience serious adverse outcomes from exposure to PM10, including premature mortality and hospitalization for respiratory conditions (See Section 7.7.3.2).

As indicated in Section 7.3, many of the epidemiologic studies demonstrate associations between PM10 and the risk of premature mortality. The extent of early mortality or life shortening may be from days to years. Although it is possible that associations between PM10 and adverse health effects may occur throughout the range of concentrations reported in each study, these occurrences are more likely when particle levels are elevated. Therefore, for purposes of these recommendations, the staff has identified the mean PM10 concentration in a given study as representing the most likely minimum effects level. This approach is consistent with that taken in the recommendation for the California 24-hour standard for sulfur dioxide. At higher mean concentrations however, the probability increases that adverse health outcomes will occur below the mean, in contrast, as concentrations decrease, the associated risks incorporate a larger range of uncertainty (see Section 7.3). In view of the current state of the science, it is not possible to identify specific levels at which no PM-related adverse effects will occur; however, the strength of the association of interest in any given study is likely to be greatest at the mean PM concentration.

Analyses of mortality (summarized in Sections 7.3 and 7.4, Table 7.1 and Figure 7.1) and morbidity (summarized in Sections 7.5 and 7.6) demonstrate that numerous epidemiological investigations have found associations of adverse health effects with PM10 when the long term (i.e., months to years) study mean concentrations are at or below the annual average standard of 30 $\mu g/m^3$. Both of the studies reporting associations between long-term exposure and mortality have mean concentrations of PM10 or its equivalent at or below the current annual average (Pope et al., 1995; Dockery et al., 1993). In the report by Dockery et al. (1993), the long-term average for PM10 ranged from 18 to 46.5 $\mu g/m^3$ in the six cities studied, with an overall mean of 30 $\mu g/m^3$. A stronger association was found for PM2.5, which ranged from 11 to 29.6 $\mu g/m^3$, in which the overall mean concentration was 18 $\mu g/m^3$. Likewise, Pope et al. (1995) reported effects from PM2.5 in the analysis of the American Cancer Society cohort, with an overall study mean of 18 $\mu g/m^3$. If the ratio of PM2.5 to PM10 is approximately 0.65, as it was in many urban areas included in the American Cancer Society study, this would convert to a PM10 average of about 28 $\mu g/m^3$. Therefore, it appears that the current annual ambient standard does not incorporate an adequate margin of safety against the occurrence of mortality associated with long-term exposures.

Although numerous epidemiological studies have demonstrated small, but consistent, relationships between health outcomes and daily variations in PM concentrations, the impacts associated with the underlying chronic exposure cannot be separated from the health effects attributed to daily PM10 or PM2.5 exposures. In other words, the daily peaks are superimposed on this underlying chronic exposure. The notion that chronic exposures exert a dominant influence on health outcomes is reinforced when one examines the mortality risks associated with daily versus chronic exposure. Most of the time-series studies demonstrate a 0.5 to 1% increase in total mortality per 10 $\mu g/m^3$ change in PM10 (Section 7.3). In contrast,

based on the American Cancer Society cohort study, the estimated mortality effect of chronic PM10 exposure is in the range of four to seven percent per 10 $\mu g/m^3$ change in the long-term average of PM10 (Pope et al. 1995; Section 7.4). These results suggest that longer-term exposures (i.e., several days to several years) account for most PM10-related mortality.

An additional complication is that, over time, the average daily PM10 concentration in a given location will be similar to the annual average PM10 concentrations. While relationships between health outcomes and daily exposure measurements can still be identified through time-series analysis, it is not possible to disentangle the influence of low-level chronic exposures with published data. Therefore, assessing the impact of occasional low-level PM peaks (e.g., at or below the level of the current 24-hour average) becomes problematic. Nonetheless, recognizing the limitations of the existing epidemiological data, the literature suggests that, when long-term mean PM10 concentrations are within the ranges reported in the published literature, it is possible to document a variety of adverse health outcomes in relation to day-to-day PM fluctuations.

Long-term mean PM levels near and below that of the current ambient California 24-hour standard have been consistently linked with respiratory symptoms and exacerbations of asthma in children. Although there are a few studies linking infant mortality to ambient PM, it is not clear, based on existing data, whether infants and children are more or less susceptible to PM-associated premature mortality than older adults with chronic heart and lunch disease. For example, it is possible that children who die of sudden infant death syndrome may have physiological abnormalities that render them unusually susceptible to the effects of PM; however, the database of published studies is too sparse for causal inference. As indicated in Section 7.7.3.2, most studies of infant mortality consist of either: (I) cross-sectional study designs, in which statistical control for all potential confounders is difficult and causal inference problematic, or (ii) time-series studies conducted in cities outside of the United States in which the PM levels are much greater than in California. In the latter group of studies, factors related to infant nutrition, health care and exposures may not be generalizable to the United States. Thus, given the current state of knowledge, it is uncertain whether infants and children represent an additional susceptible subpopulation with respect to air pollution-associated mortality at current ambient concentrations of PM. However, childhood respiratory morbidity does appear to be consistently linked with different measures of PM, within the same concentration ranges as those associated with mortality in adults with chronic heart and lung disease (See Sections 7.3 and 7.5).

The voluminous published data suggest that together, the current PM10 AAQSs are probably not adequately protective of public health particularly for the elderly and individuals with pre-existing heart or lung disease. From the perspective of public health protection, the principal shortcoming appears to be chronic PM exposures. The quantitative benefits assessment (Section 7.10) suggests that significant mortality and morbidity benefits will result from reducing population exposures to PM10.

7.11.2 Recommended Pollution Indicators

The scientific evidence suggests a need for standards to encompass fine particles as well as PM10. We therefore recommend that the PM10 indicator be retained and that a long-term standard for PM2.5 be promulgated as well. These recommendations are predicated on the following rationale:

- PM10 and fine particles are both associated with a wide range of serious adverse health outcomes, including premature mortality, hospitalizations, and asthma exacerbation, among others.

- Dosimetry studies indicate that both fine and coarse particles deposit throughout the respiratory tract (See Section 7.1). Fine particles are more likely to deposit in the alveolar region (or gas exchange zone) and may initiate inflammatory responses, with both local and systemic effects. Coarse particles (PM10 – PM2.5) can also deposit in significant quantities in the conducting airways and, to a lesser extent, in the gas exchange region of the lung. Moreover, multiple studies in which the health impacts of PM2.5 and coarse mode have been examined have reported adverse effects associated with both metrics.

- Particles larger than 10 μm in median aerodynamic diameter, which have limited deposition in either the alveolar or tracheobronchial region, are not likely to cause serious health impacts. Therefore, staff does not recommend an ambient air quality standard for particles larger than 10 μm.

- Ultrafine particles (particles with aerodynamic diameters between 0.001 and 0.1 μm), which can deposit in significant quantities throughout the respiratory tract, have been linked with serious health impacts, including premature mortality and asthma exacerbation. There is a small but growing toxicological database suggesting that ultrafine particles may be more toxic, on a mass basis, than fine particles of similar composition. However, there are few epidemiologic studies of ultrafine particles and findings are mixed. Therefore, there are insufficient data available to judge whether or not an ambient air quality standard for ultrafine particles is needed. Staff does not recommend an ambient air quality standard for ultrafine particles at this time.

- While recent toxicological research suggests potentially important roles for transition metals (e.g., iron, nickel, or vanadium) and PM-associated organic compounds in PM toxicity, there is insufficient evidence to develop ambient air quality standards for metals or any other specific chemical constituents of PM10 or PM2.5, with the exception of sulfates (see below). Therefore, staff does not recommend promulgating any other ambient air quality standard for any specific constituent of either PM10 or PM2.5. Ambient concentrations of most of the identified fine particulate constituents of potential concern, including sulfates, particulate acids, metals, and organic compounds, will be reduced by control strategies targeting PM10 and PM2.5 mass.

- Serious health effects have been associated with exposure to ambient sulfates, particularly in areas rich in strongly acidic sulfates, such as the eastern United States and Canada (See Sections 7.3, 7.4, 7.5 and 7.6). The results of such studies, however, have not been as consistent as for PM10, PM2.5 or the coarse fraction. Some studies (Gwynn et al., 2000) suggest that particle-associated hydrogen ion (H^+) and strong acidic sulfates are associated more with respiratory effects than other particle metrics, including PM10. However, in other studies, sulfates are highly correlated with the fine mode in which they predominantly occur, such that independent effects of these correlated co-pollutants cannot be reliably estimated. In a third set of studies, no association was reported for sulfates or strong particle acidity, while associations were found for PM10 (for example, Lippmann et al., 2000, Schwartz et al, 1994). In contrast to the results of some of the epidemiological studies, controlled exposure studies involving high levels (up to 1,000 $\mu g/m^3$) of strongly acidic sulfates have demonstrated little, if any, effect on volunteer subjects, including those with asthma (e.g., Aris et al. 1991). Though daily sulfate excursions in epidemiological studies have been linked with a variety of adverse health events, the nature of the study data does not allow for segregation of outcomes related to chronic low-level exposure from those associated with acute (daily) elevations in sulfate concentrations. Thus, though the mean concentrations of some multi-year studies are lower than the current 24-hour sulfate standard in California (Burnett et al., 1994; Gwynn

et al., 2000), these do not directly address the adequacy of the current 24-hour sulfate standard because it is difficult to separate the impact of a single 24-hour exposure. In this light, staff believes that the current scientific database is insufficient to use for revision of the existing sulfate standard.

In California, acidic sulfates (principally sulfuric acid and ammonium sulfate) constitute a small fraction of the PM mass relative to the areas in which sulfates have been found to be associated with adverse health impacts. For instance, in Long Beach, where the fixed-site monitor consistently shows the highest sulfate levels in the South Coast Air Basin, sulfates constitute about 13% of PM10 mass and 22% of PM2.5 mass on an annual basis, and about 16% of the maximum 24-hr PM10 mass (15 $\mu g/m^3$ sulfates/93 g/m3 PM10) and 21% of the maximum PM2.5 mass (13 $\mu g/m^3$ sulfates/61 μg/m3 PM2.5), respectively. In the San Francisco Bay Area and in Bakersfield, the percentages are much lower (California Acid Deposition Monitoring Program, 1994). In the ongoing Children's Health Study in Southern California, data on sulfates have been collected, but not yet analyzed as predictors of children's respiratory morbidity or lung function growth and development. According to ARB staff, these data should be analyzed over the next couple years.

In general, sulfates detected in California are less strongly acidic than those commonly found in the eastern United States and Canada. Though a time-series study linked sulfate concentrations in 1978-79 in Azusa, California with respiratory symptom reporting in adults, ambient levels during that study period exceeded the standard (Ostro et al., 1993). Sulfate concentrations in California have been lower, typically far lower, during the past few years than the level of the existing standard. Although a mortality time-series study undertaken in Santa Clara County (1989-1996) involving very low 24-hour average sulfate values (mean = 1.8, range 0-7.9 g/m^3) suggests an association with daily respiratory mortality, staff believes this finding can be attributed principally to the strong covariation of sulfates with PM2.5 (Fairley, 1999). Based on an assessment of current scientific evidence and ambient air quality data, staff believes that exposures to sulfates in California do not appear to pose health risks distinct from or greater than those associated with exposures to particulate matter generally. In view of the mixed evidence in the sulfates health effects literature, the paucity of recent data examining sulfates and health in California, the low likelihood of health risks in relation to ongoing trends in sulfate emissions and ambient levels, staff recommends the current standard be retained until the next review of the PM standard.

In the review of the adequacy of the California AAQS to protect public health mandated by the Children's Environmental Health Protection Act (ARB 2000), much of the evidence regarding the health impacts of sulfates was based on considerations of the PM epidemiology. Revisions of California's PM standards as recommended (below) will likely further reduce sulfate concentrations. In addition, based on discussions with ARB staff, the differences in sulfate composition and levels between California and the eastern United States are sufficient for OEHHA staff to recommend further studies in California prior to a full review of the sulfate standard. In particular, OEHHA staff recommends analysis of the sulfate data in relation to health indicators in the Children's Health Study, as well as time-series analyses of health outcomes and daily sulfate data being collected at the two California particulate matter Supersites in Los Angeles and Fresno. OEHHA recommends that ARB ensure that these analyses be conducted in such a manner as to provide optimally useful data for a full review of the sulfate standard.

- PM2.5 can infiltrate directly into residences, with greater penetration than the coarse fraction, and therefore individuals are likely to have more consistent indoor exposure to ambient PM2.5 than to the coarse fraction. Nevertheless, the coarse fraction also

demonstrates substantial indoor infiltration, particularly in older buildings, or those in which windows or doors are kept open. Evidence from studies in California, indicate that 75% of indoor PM2.5 and 65% of indoor PM10 may originate outdoors (Ozkaynak et al., 1996b; see Chapter 6). Therefore, outdoor, ambient concentrations of PM2.5 and PM10 will play a significant role in total, personal exposure.

- Fine and coarse particles, in general, originate from different sources and have different lung penetration and deposition characteristics, but are both linked to adverse health effects. In most California cities, mobile sources are a significant source of PM10. In these cities, there are strong daily correlations between PM2.5 and PM10 throughout much of the year, such that a substantial fraction of PM10-associated health impacts can be reasonably ascribed to PM2.5.
- In contrast, PM2.5/PM10 ratios are lower in many parts of California than those observed nationally (Chapter 6). In some parts of the state, particularly in the inland air basins in Southern California, high PM10 concentrations are driven by the coarse mode. However, at this time, the current research database regarding coarse particles' health impacts is not as well developed as that for PM10. Therefore, staff recommends that PM10 standards be used as a basis for protection from exposure to coarse particles.

Taking into account all of the above factors, therefore, staff recommends the Air Resources Board promulgate new annual standards for PM10 and PM2.5, while retaining the existing 24-hour standards for PM10 and sulfates.

7.11.3 Averaging Times and Forms

The current PM10 AAQSs for California include both an annual standard based on the geometric mean concentration, and a 24-hour averaging time, not to be exceeded during the calendar year. These joint standards were developed to protect the public from both long-term and short-term exposures. Studies published since the California PM10 AAQSs were developed in the early 1980s support earlier findings and report associations between adverse health outcomes and both long-term (i.e., a year or longer) and short-term (i.e., from less than one day to several months) exposure to both PM10 and fine particles. Therefore, staff proposes standards using annual averages for PM10, PM2.5 and sulfates, and a shorter-term average for PM10. The foundations for the annual averages are relatively straightforward, as explained in the subsections below. Identifying a shorter-term average based on the existing epidemiological database is somewhat more difficult conceptually due principally to the intermingling of effects related to chronic and acute exposure, as described in Section 7.11.1, above.

While there is evidence of health effects associated with other averaging times (e.g., 4-hour and multi-year averages), staff believes that proposed averaging times will provide a satisfactory basis for setting PM standards and directing subsequent pollution control efforts.

Attainment of the annual standards described below will shift the current distributions of PM10, the coarse fraction, and PM2.5 to levels substantially lower than currently exist. Therefore, 24-hour averages of ambient concentrations of these particle measures will also decline. This implies that the current 24-hour average standard for PM10 should, unlike today, only occasionally be exceeded in most air basins. However, data developed by ARB staff indicate that even if the proposed annual PM10 standard is attained, some parts of California will sporadically experience PM10 excursions well above the current standard. Therefore, short-term standards will function primarily to address intermittent seasonal exceedances (e.g., from residential wood combustion during the winter holiday season or prolonged

summer temperature inversions) that might occur in air basins otherwise in attainment with the annual averages.

For the annual averages, OEHHA staff recommends using the arithmetic rather than the geometric mean because the former is: (1) more directly related to cumulative exposure; (2) more sensitive to repeated peak concentrations; and (3) more consistent with other annual standards.

7.11.4 Recommended Concentrations

Although individual epidemiologic studies are subject to some uncertainty, particularly with respect to exposure assessment, the overall body of evidence (including toxicologic, dosimetric and human clinical studies, in addition to the epidemiological investigations) particularly the consistency and coherence of results, provides compelling evidence of causal relationships between exposure to ambient PM and a variety of adverse health outcomes (See Section 7.9). These studies provide a sound, scientific basis for the establishment of standards for both PM2.5 and PM10.

While several indicators of morbidity have been associated with exposures to ambient PM, including hospital admissions, emergency room visits, exacerbation of asthma, work loss, school absenteeism, bronchitis and respiratory symptoms, and changes in lung function, the choices of levels for the annual average standards set forth below are based primarily on studies of mortality. This is clearly the most definitive and serious of all the health events associated with exposure to PM. The mortality exposure-response relationship appears to be linear, at least for cardiorespiratory deaths, with no evidence of a threshold of effect within the range of the long term means of 24-hour average PM10 concentrations reported in the daily mortality studies (i.e., Daniels et al., 2000). PM-associated mortality has been observed at long-term average ambient concentrations comparable to those at which morbidity outcomes have been detected in other populations (See Sections 7.3 – 7.6), which suggests that it would be reasonable to base the standards principally on studies involving the most serious outcome. To our knowledge, there is no evidence that morbidity effects would occur at PM concentrations lower than those associated with increased risks of mortality. This may be due to the different populations at risk examined in the various studies. That is, associations between 24-hour averages and mortality have been detected primarily in the elderly who have a high prevalence of chronic cardiac and respiratory disease. In contrast, time-series or panel studies of children, who are not at high risk of mortality, have examined a variety of respiratory morbidity outcomes in relation to daily changes in PM. Though the initiation of biological reactions may overlap (i.e., airway and alveolar inflammation), the downstream pathophysiological consequences will vary. As there does not appear to be a gradient of exposure concentrations related to increasing health outcome severity, standards premised on providing protection against mortaility should also, *a fortiori*, protect the public, including infants and children, against the occurrence of morbidity outcomes.

To the extent that the annual standards for PM10 and PM2.5 are attained, the distributions of 24-hour and other short-term averages of PM10 and PM2.5 will shift downward markedly throughout the year. The likelihood of adverse health events occurring after acute exposures will also therefore be substantially reduced. Nevertheless, there may well be areas that will attain the annual PM standards, yet still experience seasonally high PM excursions associated, for instance, with prolonged winter air stagnation combined with residential wood combustion or with summer temperature inversions. The plethora of time-series and panel studies cited in this document make it clear that short-term elevations of PM are associated with increased morbidity and mortality, though again, the impacts of the ongoing chronic PM exposure have not been identified. Therefore, though downward revisions to the annual PM

standard will enhance protection of the health of the public, including infants and children, it is appropriate to limit shorter-term PM exposures.

7.11.4.1 Annual Standard for PM10

Considering the weight of evidence from the literature reviewed in prior sections, staff recommends the annual average standard for PM10 should be revised from 30 to 20 $\mu g/m^3$. Consideration of an annual standard at this level would place significant weight on the studies of mortality related to long-term PM exposure using the Harvard Six-Cities data (Dockery et al. 1993) and the American Cancer Society cohort (Pope et al., 1995), both reanalyzed by Krewski et al. (2000). In the study by Dockery et al. (1993), the long-term average for PM10 ranged from 18 to 46.5 $\mu g/m^3$ in the six cities, with an overall mean of 30 $\mu g/m^3$. Visual inspection of graphs of this study's results suggests a continuum of effects down to the lowest levels, with no evidence for a threshold, (although it would be difficult to ascertain a threshold graphically in this set of six data points corresponding to the six cities). However, the city with the lowest long-term average PM10 concentration (Portage, Wisconsin) was, for purposes of analysis, designated as the reference category, against which the other cities were compared. In other words, it was assumed in the analysis that there was no increase in risk in this city. Thus, it would *not* be appropriate to infer, for standard-setting purposes, that PM-related effects on mortality occurred at the long-term mean PM10 concentration of 18 $\mu g/m^3$ in Portage. In addition, while there appears to be a graphic exposure-response relationship by city, no clear increase in the risk of mortality is evident in Topeka, KS (which had a long-term annual PM10 concentration of 26.4 $\mu g/m^3$) relative to Portage. Finally, the relevant periods of exposure associated with long-term effects are unknown (other than those likely to be associated with short-term exposures within each year). In the absence of better information, it is reasonable to select the mean long-term PM10 level as a starting point for recommending the annual standard. In the Six-Cities study, the mean long-term PM10 level was 30 $\mu g/m^3$.

Likewise, Pope et al. (1995) reported effects on mortality associated with PM2.5, but not PM10, in the analysis of the American Cancer Society cohort, with an overall PM2.5 study mean of 18 $\mu g/m^3$. The recent re-analysis of the ACS study also suggests effects of PM2.5, but not PM10, related to long-term exposures (Krewski 2001). If one assumes that fine particles are driving the associations between PM and mortality in the ACS study, and that the ratio of PM2.5 to PM10 is about 0.65 for most of the urban areas included in that study (see Chapter 6), this would convert to an overall long-term average PM10 concentration of 28 $\mu g/m^3$.

Several investigations, including the Children's Health Study (McConnell et al. 1999) and the Harvard Six-Cities Study (Dockery et al., 1989), have also reported associations between long-term PM exposures and morbidity outcomes, including bronchitis, exacerbation of asthma, and reductions in lung function (See section 7.6). In these studies, the long-term (one- or multi-year) mean PM10 concentrations ranged from about 21 to 35 $\mu g/m^3$. Some of the morbidity studies, however, may be capturing the effects of exposure to multiple pollutants. For instance, in the Children's Health Study, the associations of adverse health outcomes with PM10 and PM2.5 could not be statistically disentangled from the co-pollutants NO_2 and acid vapors. Therefore, selection of a target concentration of 20 $\mu g/m^3$ puts greater likelihood on a PM-specific effect in these morbidity studies, and provides a margin of safety, assuming that there may be interactions among co-pollutants.

As noted above, the epidemiological studies of daily exposure and mortality have reported mean or median PM10 concentrations from 14 to 115 $\mu g/m^3$ (see Table 7.1 and Figure 7.1). However, the degree of uncertainty regarding the results generally decreases as the average or median concentration increases. As can be seen in Figure 7.2, almost all of the studies

with means or medians below 25 μg/m^3 have point estimates suggesting an association with PM10, but the confidence intervals include the null value, indicating weaker associations that are more uncertain. The annual averages of these short-term exposure studies are relevant, since effects are observed throughout a wide range of exposures and not only at the extreme values. In addition, some of the PM-associated mortality captured in the cohort studies above would include the modest increments in the short-term risks described in the time-series studies, recognizing that larger long-term increments in risk appear to be related more to chronic than to short-term exposures. Finally, all of the time-series studies conducted at these lower concentrations were undertaken outside California and the United States. Studies more relevant to California (i.e., those conducted in California or other parts of the United States) reported long-term PM concentrations in the range of 25 to 35 μg/m^3 (see Table 7.1). Consideration of a standard of 20 μg/m^3 would, therefore, provide a margin of safety by placing significant weight on some of the time-series studies conducted outside of California and the U.S. This recognizes the generalizability of the results of these studies, although the sources and mix of PM constituents, the underlying population health characteristics, and the exposure patterns may differ from those in California. A standard set at 20 μg/m^3 would protect against mortality effects related to long-term exposure in adults and morbidity effects (such as acute bronchitis in children). The quantitative benefits assessment suggests that attainment of this standard could result in the avoidance of an estimated 6,500 (95% CI=3,200-9800) cases of premature mortality per year associated with the difference between this proposed level and the current annual averages of ambient PM10 concentrations throughout California (a population-weighted average exposure of 33.1 μg/m^3).

7.11.4.2 24-hour Average for PM10

Staff recommends that the 24-hour average for PM10 of 50 μg/m^3, not to be exceeded, be retained. This standard would offer protection primarily against peak concentrations of both fine and coarse particles in areas that otherwise attain the annual standards for PM10 and PM2.5. For many urban areas in California, attainment of the annual standards will mean infrequent PM excursions, which would typically be associated with seasonal air stagnation. Thus, the 24-hour standard would is intended to prevent occasional elevated PM10 levels. Staff believes that the existing PM10 24-hour standard proscribing any single day concentration above 50 μg/m^3, in concert with attainment of the annual average standards for PM10 and PM2.5, provides substantial protection of public health, including that of infants and children, as described below.

The 24-hour PM10 standard was first promulgated in California in 1983, based primarily on an analysis of daily mortality in London in relation to changes in PM. At that time, there were no epidemiological studies in which PM10 had actually been measured. Rather, critical PM10 concentrations had estimated from other PM metrics, including TSP and British Smoke. Since then, a voluminous literature has appeared linking fluctuations in short-term or daily measurements of PM10 with a variety of adverse health outcomes, as reviewed in Sections 7.2, 7.3 and 7.5. Complemented by recent toxicological and controlled human exposure studies, the epidemiological foundation linking variations in ambient PM10 and daily morbidity and mortality has been firmly established.

Nonetheless, translating the results of these epidemiological studies into a short-term standard remains problematic. As noted in prior sections, multi-city analyses in Europe and the United States suggest exposure-response relationships between daily variations in ambient PM10 and fluctuations in cardiopulmonary mortality and other health effects that are essentially linear and without an observable threshold. To the extent that this is an accurate characterization of PM10-mortality associations, and that the latter represent causal

relationships, there is little guidance on where to draw a "bright line" in recommending a short-term standard. Moreover, in time-series studies it is difficult to identify and separate the influence of chronic low-level exposures in contributing to individuals' susceptibility to daily PM elevations. Cumulative exposures over several days or longer, rather than during a single 24-hour period, may represent a more relevant time frame of exposure. Consistent with this hypothesis, numerous epidemiological studies report morbidity or mortality effects of greater magnitude associated with multi-day moving averages compared with single-day lags (Hajat et al., 2001; Schwartz, 2000b; Schwartz et al., 1993; Pope et al., 1992).

Recognizing the limitations of the epidemiological data available for standard-setting purposes, OEHHA recommends retention of the 24-hour standard in consideration of the following factors: (1) the apparent linearity of dose-response; (2) the greater uncertainty of effects at the lower concentrations; (3) the paucity of epidemiological data documenting the impact of a single 24-hour exposure at low ambient (i.e., non-occupational) concentrations; (4) the dominance of the effects associated with chronic exposures and the impact of chronic exposure on the response to short-term elevations in PM concentration; (5) the likelihood of effects occurring at concentrations above 50 $\mu g/m^3$ and (6) the interrelationships of alternative averaging times.

Linearity of Dose-Response

As discussed above (Section 7.3.5), time-series studies of morbidity and mortality indicate that the exposure-response relationships for 24-hour average PM exposures are linear and show no evidence of a threshold. The latter observation makes it difficult to identify where a "bright line" representing a single-day 24-hour PM10 standard should be drawn. The historic rationale for a 24-hour standard was the presumption that significant health effects occurred only on high concentration, "episodic" days or that high pollution days generated disproportionately greater and more severe adverse health outcomes. In general, the notion that episodic peaks alone are responsible for adverse effects ignores the potential role of chronic low-level exposures, which may predispose individuals towards greater susceptibility to elevated PM concentrations. In addition, there is little, if any, evidence that the exposure-response relationship becomes steeper at higher ambient concentrations; rather, the data generally indicate a linear exposure-response relationship.

Greater Uncertainty at Lower Concentrations

Epidemiological studies of short-term exposure and mortality have reported mean or median PM10 concentrations ranging from 14 to 115 $\mu g/m^3$ (see Table 7.1 and Figure 7.1). As can be seen in Figures 7.2 and 7.3, however, greater uncertainty about the effects exists as one moves to studies with lower concentrations. The greater uncertainty may be due to fewer health impacts associated with exposure to lower concentrations as well as other factors, including errors in exposure measurement, confounding by co-pollutants, and the chemistry of the particle mixture. Other uncertainties related to extrapolating the epidemiological findings from many of the daily exposure studies to California may result from differences in factors such as weather, housing stock, and population characteristics. Therefore, retention of the existing 24-hour standard acknowledges the uncertainty in applying the underlying studies with relatively low PM10 levels, particularly those conducted in other countries, to urban and suburban populations in California.

Impact of Single 24-Hour Exposures at Low Concentrations

Exposures of 24-hours duration occur "on top of" consistent chronic low-level exposures to PM. The effects of long-term exposure to PM, as described in Section 7.4, have been

documented in several carefully conducted studies using a prospective cohort design. These studies incorporate effects associated with both short-and long-term exposures (although they may not include all of the impacts associated with mortality displacement). Basically, for these study effects to be observed, individuals must be continually moving into a "risk pool" from a non-risk or lower-risk status over time. Long-term exposure to PM subjects people to an increased risk (i.e., moves then into the "risk pool") of mortality from cardiovascular disease, whether or not their deaths are ultimately associated with a recent "acute" exposure to PM (Schwartz, 2001a; Kunzli et al., 2001). While acute daily exposures appear to exert an independent effect on mortality and morbidity, the influence of a single 24-hour exposure at a concentration relevant to the PM standards, absent any other exposure to PM, has not been (and probably cannot be) determined epidemiologically. This would require observance of weeks or months of exposure to very low background levels of PM followed by a single day peak exposure. Even for individuals exposed experimentally in chamber studies, prior exposure to ambient PM cannot be discounted. Therefore, it is difficult to completely isolate the impacts of short-term elevated PM levels from chronic background exposures. In addition, as reviewed above, there is evidence that multi-day PM10 exposures are, at least in some studies, associated with greater risks than single-day exposures.

Importance of Impacts of Chronic Exposure

Our quantitative benefits assessment (Section 7.10) as well as similar efforts undertaken recently by the U.S. EPA (U.S. EPA, 2000) indicates that the total health impacts of PM are dominated by mortality associated with long-term exposure. In addition, effects on adult cases of bronchitis and childhood acute bronchitis, both associated with longer-term exposure to PM, are significant as well. Therefore, from a public health perspective, one should focus control strategies on reducing the entire distribution of PM concentrations, which would also lower the number of peak days. Formulating a short-term index consistent with the annual average is a rational way to approach the issue of limiting peak exposures that might still occur even when the annual average PM standard is attained.

Relationship of Recommended 24-hour and Annual PM10 Standards

As discussed in Chapter 6, ARB uses the Expected Peak Day Concentration (EPDC) in determining the "design value" for the 24-hour standard. The development of the EPDC uses a statistical model of the highest 20% of the daily values from the previous three years, making it relatively robust with respect to fluctuations in daily meteorological conditions. Specifically, the index will not be unduly influenced by any single day, and exceptional events such as forest or urban fires can be excluded. We conducted an analysis to determine the relationship between the EPDC and the annual average of 20 $\mu g/m^3$, the most health-protective end of the range proposed above. This analysis identified the single day peak exposure concentration that is consistent, given the current statewide distributions of PM10, with an annual average of 20 $\mu g/m^3$.

Using data from 144 sites around the state, a linear regression model was run relating the EPDC to the annual average for each site. The regression model generated an r^2 of 0.72 and indicated that statewide, the EPDC associated with a 20 $\mu g/m^3$ annual average is 48 $\mu g/m^3$ which accords quiet closely with the existing standard. For the South Coast AQMD, representing the most populous air basin in the state, the predicted EPDC is 51 $\mu g/m^3$.

Likelihood of Effects Occurring at Single Exposures Above 50 μg/m^3

As indicated by Table 7.1, several studies with study means in the range of 15 to 30 μg/m^3 PM10 demonstrate associations between daily exposures and mortality. However, as indicated above, several studies at the lower concentration had confidence intervals that included an estimate of no effect; that is where the null hypothesis of no effect could not be rejected. OEHHA staff has examined the distribution of peak concentrations (i.e., 95th percentiles or maximum 24-hour concentrations) when they were provided in the time-series mortality studies reporting study mean concentrations of less than 30 μg/m^3. Many of these studies have peak values close to or above 50 μg/m^3. Keeping peak concentrations below 50 μg/m^3 will not assure the absence of health impacts. However, peak concentrations below this level are consistent with a distribution of PM10 in which the likelihood of mortality effects are less certain. Therefore, it is reasonable, from a public health perspective, to recommend a goal of preventing days when the 24-hour average concentration exceeds 50 μg/m^3.

In summary, while it is difficult to determine the effects of a single 24-hour exposure from available scientific studies, the evidence suggests that minimizing or eliminating days when the 24-hour PM10 average concentration exceeds 50 μg/m^3 is a prudent public health goal. Taking into account all of the scientific evidence, and bearing in mind that the attainment of the annual average standard will significantly depress the entire PM10 distribution, preventing single day concentrations below 50 g/m^3 should afford additional public health protection. Therefore we are proposing that the 24-hour standard be retained 50 μg/m^3. Future research should focus on the implications of short-term exposures of 24-hours or less in the absence of cumulative or chronic exposures to PM10. Together, these standards should protect public health with an adequate margin of safety in the sense described in the introductory paragraphs of Section 7.11.

7.11.4.3 Annual Standard for PM2.5

Staff recommends that the annual average for PM2.5 should be 12 μg/m^3, as explained below. Consideration of a standard at this level would place significant weight on the long-term exposure studies using the ACS and Harvard Six-Cities data (Dockery et al., 1993; Pope et al., 1995; Krewski et al., 2000). In both studies, robust associations were reported between long-term exposure to PM2.5 and mortality. The mean PM2.5 concentration was 18 μg/m^3 (range of 11.0 to 29.6 μg/m^3) in the Six-Cities study and 18.2 μg/m^3 (range of 9.0 to 33.5 μg/m^3) in the ACS study. Thresholds were not apparent in either of these studies, although the relevant period(s) and pattern(s) of exposure could not be ascertained. If we assume, as in the PM 10 standards considered above, that health effects are more likely to be observed when concentrations are at or above the mean or median PM2.5 levels, rather than at lower levels, then a reasonable starting point for considering an annual PM2.5 standard would be 18 μg/m^3.

Targeting a long-term mean PM2.5 concentration of 12 μg/m^3 would also place some weight on the results of multiple daily exposure studies examining relationships between PM2.5 and adverse health outcomes (Table 7.2). These studies have long-term (three- to four-year) means in the range of 13 to 18 μg/m^3. It should be noted however, that many of these epidemiological investigations were conducted outside California, and may not be representative of exposures or population characteristics here. A standard set at 12 μg/m^3, well below the means of the major cohort mortality studies, would provide additional protection against mortality in adults associated with long-term exposure, as well as against a variety of morbidity effects in children (described in Section 7.6, above). In the opinion of OEHHA staff, an annual PM2.5 standard of 12 μg/m^3 would be likely to provide adequate

protection of public health, including that of infants and children, against adverse effects of long-term exposure.

The quantitative risk assessment suggests that attainment of this standard could result in a reduction of 6,500 cases (95 percent CI 3,200 – 9,800) of premature mortality per year associated with the current annual averages of ambient PM2.5 concentrations in the diverse air basins of California (approximately 18.5 $\mu g/m^3$, as reported in Chapter 10.

7.11.4.4 24-hour Standard for PM2.5

Staff does not recommend a 24-hour average standard for PM2.5 at this time. Staff recognizes that PM2.5 exposures can have significant, short-term health impacts. While effects resulting from long-term exposure to fine particles are evident from the prospective cohort studies, there are fewer studies on effects from shorter exposures. As indicated in the review of the few studies of daily mortality in relation to ambient PM2.5, a consistent differential in the acute effects of fine versus coarse particles is not evident. In addition, data from California indicate that for most urban areas, days with high PM10 concentrations are associated with high PM2.5 concentrations. Therefore, the 24-hour average PM10 standard should provide control for 24-average PM2.5 peaks as well. During the next cycle of review of the PM standards, there should be a larger database of PM2.5 studies to evaluate as the basis for a potential short-term fine particle standard. At that time, staff will again evaluate the potential for short-term PM2.5 standards.

7.11.4.5 24-hour Standard for Sulfates

Staff recommends that the 24-hour average for sulfate of 25 $\mu g/m^3$, not to be exceeded, be retained. Serious health effects have been associated with exposure to ambient sulfates, particularly in areas rich in strongly acidic sulfates such as the eastern United States and Canada. The results of such studies however, have not been as consistent as those for PM10, PM2.5, or the coarse fraction. In addition, though daily sulfate concentrations have been linked with a variety of adverse health events in epidemiological studies, the nature of the study data does not allow for segregation of outcomes related to chronic low-level exposure from those associated with daily elevations in sulfate concentrations.

In California, acidic sulfates (principally sulfuric acid and ammonium sulfate) constitute a small fraction of the PM mass relative to the areas in which sulfates have been found to be associated with adverse health impacts. Sulfate concentrations in California have been far lower during the past few years than the level of the existing standard. Based on an assessment of current scientific evidence and ambient air quality data, staff believes that exposures to sulfates in California do not appear to pose health risks distinct from or greater than those associated with exposures to particulate matter generally. In view of the mixed evidence in the sulfates and health in California, the low likelihood of health risks in relation to ongoing trends in sulfate emissions and ambient levels, staff recommends that the current standard be retained until the next review of the PM standard, if not earlier.

7.11.4.6 Other Recommendations

In light of the adverse health effects observed at current ambient concentrations and the lack of a demonstrated threshold, staff further recommends: (1) that in any air basin in California that currently attains the ambient air quality standards, for either PM10 or PM2.5, the air quality should not be degraded from present levels; and (2) that the ARB, in consultation with local air quality management districts, establish a goal of continued reductions in PM10 and PM2.5 concentrations over time. We further recommend that the standards be revisited within five years, in order to re-evaluate the evidence regarding the health effects associated with particle size, chemistry, and concentration.

7.11.5 Summary of Recommendations

- Revise the current PM10 annual average standard from 30 to 20 $\mu g/m^3$. Revise the averaging method to an annual arithmetic mean from the current annual geometric mean. Based on current evidence, there are compelling reasons to be concerned about significant adverse health effects associated with exposures occurring at or below the existing standard.
- Retain the 24-hour standard for PM10 at 50 $\mu g/m^3$, not to be exceeded.
- Establish an annual average standard for PM2.5 of 12 $\mu g/m^3$, given growing evidence from epidemiological and toxicological studies of significant toxicity related to this size fraction of PM. Establish the annual PM2.5 standard as an annual arithmetic mean.
- Retain the current 24-hour average standard of 25 $\mu g/m^3$ for sulfates.

General Staff Conclusions Regarding Air Quality Degradation

- For any air basin in California that currently attains the ambient air quality standards, for either PM10 or PM2.5, that air quality should not be degraded from present levels.
- Establish a goal of continued reductions in PM10 and PM2.5 concentrations over time.
- The standards be revisited within five years, in order to re-evaluate the evidence regarding the health effects associated with particle size, chemistry, and concentration.

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Table 7.1. Summary of Short-term PM10 Mortality Studies

ID	City	Country	Author(s), Year Published	Age Group*	Mean (BOLD=Median)	% Daily Mortality Increase per 10mg/m^3	# of obs
1	Stockholm	Sweden	Katsouyanni et al., 2001	1	**14**	0.39	2555
2	Portage, WI	US	Schwartz et al., 1996	1	**18**	0.70	1436
3	Sydney	Australia	Morgan et al., 1998	1	18	0.95	1795
4	Ottawa	Canada	Burnett et al., 2000	1	20	1.45	433
5	Edinburgh	Scotland	Prescott et al., 1998	1	21	0.10	1460
6	Birmingham	England	Katsouyanni et al., 2001	1	**21**	0.28	1825
7	Vancouver	Canada	Burnett et al., 2000	1	22	1.46	565
8	Paris	France	Katsouyanni et al., 2001	1	**22**	0.43	2190
9	Helsinki	Finland	Katsouyanni et al., 2001	1	**23**	0.32	1460
10	Edmonton	Canada	Burnett et al., 2000	1	23	1.28	508
11	Buffalo-Rochester, NY	US	Gwynn et al., 2000	1	24	2.33	175
12	Boston, MA	US	Schwartz et al., 1996	1	**25**	1.20	1140
13	London	England	Katsouyanni et al., 2001	1	**25**	0.69	1825

*Age Group
1 = All ages
2 = 65 yrs and older
3 = under 65 years

ID	City	Country	Author(s), Year Published	Age Group*	Mean (BOLD =Median)	% Daily Mortality Increase per 10mg/m^3	# of obs
15	Birmingham	England	Wordley et al , 1997	1	26	1.10	730
16	Winnipeg	Canada	Burnett et al., 2000	1	26	0.35	538
17	Toronto	Canada	Burnett et al., 2000	1	26	0.67	889
18	Topeka, KS	US	Schwartz et al., 1996	1	27	-0.50	1432
19	Montreal	Canada	Burnett et al., 2000	1	27	0.51	853
20	Basel	Germany	Katsouyanni et al., 2001	1	**28**	0.41	2190
21	Helsinki	Finland	Ponka et al., 1998	3	**28**	3.45	2555
22	Minneapolis, MN	US	Braga et al., 2000	1	28	1.34	2920
23	St. Louis, MO	US	Dockery et al., 1992	1	28	1.50	311
24	Zurich	Switzerland	Katsouyanni et al., 2001	1	**28**	0.42	2190
25	London	England	Bremner et al., 1999	1	29	0.26	1095
26	Kingston-Knoxville, TN	US	Dockery et al., 1992	1	30	1.60	330
27	St Louis, MO	US	Schwartz et al., 1996	1	**31**	0.60	1375
28	Detroit, MI	US	Lippmann et al., 2000	1	31	0.86	490
29	Windsor	Canada	Burnett et al., 2000	1	31	2.88	850
30	Knoxville, TN	US	Schwartz et al., 1996	1	**32**	0.90	1481

*Age Group

1 = All ages

2 = 65 yrs and older

3 = under 65 years

ID	City	Country	Author(s), Year Published	Age Group*	Mean (BOLD =Median)	% Daily Mortality Increase per 10mg/m^3	# of obs
31	Montreal	Canada	Goldberg et al., 2000	1	32	0.67	3650
32	Seattle, WA	US	Braga et al., 2000	1	32	0.52	2920
33	Ogden, UT	US	Pope et al., 1999a	1	32	1.62	2308
34	Geneva	Switzerland	Katsouyanni et al., 2001	1	**33**	-0.10	2190
35	Madrid	Spain	Katsouyanni et al., 2001	1	**33**	0.53	1460
36	San Jose, CA	US	Fairley , 1999	1	34	1.54	823
37	Chicago, IL	US	Braga et al., 2000	1	36	0.81	2920
38	Chicago, IL	US	Schwartz, 2001a	1	**36**	0.89	2190
39	Detroit, MI	US	Braga et al., 2000	1	36	0.87	2920
40	Pittsburgh, PA	US	Braga et al., 2000	1	**36**	0.84	2920
41	Provo/Orem, UT	US	Pope et al., 1999a	1	38	0.95	3687
42	Lyon	France	Katsouyanni et al., 2001	1	**39**	1.35	1825
43	Athens	Greece	Katsouyanni et al , 2001	1	**40**	1.53	1825
44	Budapest	Hungary	Katsouyanni et al., 2001	1	**40**	0.29	1460
45	Chicago, IL	US	Ito and Thurston, 1996	1	**41**	0.50	1529

*Age Group

1 = All ages

2 = 65 yrs and older

3 = under 65 years

ID	City	Country	Author(s), Year Published	Age Group*	Mean (BOLD =Median)	% Daily Mortality Increase per 10mg/m^3	# of obs
46	Salt Lake City, UT	US	Pope et al., 1999a	1	41	0.77	3700
47	Teplice	Slovakia	Katsouyanni et al., 2001	1	**42**	0.64	2920
48	Tel Aviv	Israel	Katsouyanni et al., 2001	1	**43**	0.64	2190
49	Mexico City	Mexico	Castillejos et al., 2000	1	45	1.83	866
50	Detroit, MI	US	Lippmann et al., 2000	1	45	0.34	1565
51	Steubenville, OH	US	Schwartz et al., 1996	1	**46**	0.90	1520
52	Phoenix, AZ	US	Mar et al., 2000	2	46	1.06	1095
53	Coachella Valley, CA	US	Ostro et al., 2000	1	47	0.41	3011
54	Milano	Italy	Katsouyanni et al., 2001	1	**47**	1.16	2555
55	Utah Valley,	US	Pope et al., 1992	1	47	1.47	1706
56	Birmingham, AL	US	Schwartz , 1993	1	48	1.10	1248
57	Erfurt	Germany	Katsouyanni et al., 2001	1	**48**	-0.56	1825
58	Cracow	Poland	Katsouyanni et al., 2001	1	**54**	0.13	2555
59	Rome	Italy	Katsouyanni et al., 2001	1	**57**	1.28	1825

*Age Group

1 = All ages

2 = 65 yrs and older

3 = under 65 years

ID	City	Country	Author(s), Year Published	Age Group*	Mean (BOLD =Median)	% Daily Mortality Increase per 10 $\mu g/m^3$	# of obs
60	Los Angeles,	US	Kinney et al., 1995	1	58	0.50	364
61	Barcelona	Spain	Katsouyanni et al., 2001	1	**60**	0.93	2190
62	Bangkok	Thailand	Ostro et al., 1999a	1	65	1.70	1431
63	Torino	Italy	Katsouyanni et al., 2001	1	**65**	1.05	2555
64	Prague	Czech	Katsouyanni et al., 2001	1	**66**	0.12	1795
65	Sao Paulo	Brazil	Saldiva and Bohm, 1995	2	82	1.31	365
66	Rome	Italy	Michelozzi et al., 1997	1	84	0.66	1278
67	Santiago	Chile	Ostro et al., 1996	1	115	1.13	779

*Age Group
1 = All ages
2 = 65 yrs and older
3 = under 65 years

Table 7.2. Studies of short-term exposure and daily mortality associated with fine and coarse particles

City	Country	Author(s) Year Published	Age Group*	Particle Type	Mean (Bold=Median)	% Mortality increase per 10 $\mu g/m^3$
8 Cities (Montreal, Ottawa, Toronto, Windsor, Winnipeg, Calgary, Edmonton, Vancouver)	Canada	Burnett et al., 2000	1	FP	13	1.20
				CP	13	0.71
Mexico City	Mexico	Castillejos et al., 2000	1	FP	27	1.48
				CP	17	4.07
Pittsburgh, PA	US	Chock et al., 2000	3	FP	NA	0.59
				CP	NA	0.50
Santiago	Chile	Cifuentes et al., 2000	1	FP	64	0.73
				CP	47	0.91
St. Louis, MO	US	Dockery et al., 1992	1	FP	18	1.71
Kingston, TN	US	Dockery et al., 1992	1	FP	21	2.28
Santa Clara, CA	US	Fairley , 1999	1	FP	13	3.26
				CP	11	1.77
Montreal	Canada	Goldberg et al., 2000	1	FP	18	1.93
Detroit, MI	US	Lippmann et al., 2000	1	FP	18	1.24
				CP	13	1.58
Phoenix, AZ	US	Mar et al., 2000	2	FP	13	2.22
				CP	34	1.17
Coachella Valley, CA	US	Ostro et al., 2000	1	FP	17	4.44
				CP	31	0.51
Harvard 6-Cities (Boston, Knoxville, St Louis, Steubenville, Portage)	US	Schwartz et al., 1996	1	FP	**15**	1.50
				CP	**9**	0.40
Newark, NJ	US	Tsai et al., 2000	1	FP	42	1.70

*Age Group

1 = All ages

2 = 65 yrs and older

3 = 75 years and older

Table 7.3. Summary of Studies of Short-term Exposure and Hospital Admissions

City	Time Period	Author(s), Year Published	Age Group	Endpoint	Particle Type	% Increase per 10mg/m^3 and 95% CI	Mean (Bold=Median)
London	(1992-94)	Atkinson et al., 1999	all ages	CV:	PM10	0.64 (0.18, 1.10)	29
London	(1992-94)	Atkinson et al., 1999	under 65	CV:	PM10	1.12 (0.40, 1.88)	29
London	(1992-94)	Atkinson et al., 1999	65 and older	CV:	PM10	0.50 (-0.04, 1.06)	29
London	(1992-94)	Atkinson et al., 1999	under 65	CV:IHD	PM10	1.36 (0.26, 2.54)	29
London	(1992-94)	Atkinson et al., 1999	65 and older	CV:IHD	PM10	1.00 (0.16, 1.86)	29
Toronto metro area	(1992-94)	Burnett et al., 1997b	all ages	CV:	CP	5.40 (2.20, 8.80)	12
Toronto metro area	(1992-94)	Burnett et al., 1997b	all ages	Resp:	CP	5.00 (2.08, 8.00)	12
Toronto metro area	(1992-94)	Burnett et al., 1997b	all ages	CV:	FP	2.36 (0.72, 4.08)	17
Toronto metro area	(1992-94)	Burnett et al., 1997b	all ages	Resp:	FP	3.40 (1.36, 5.52)	17
Toronto metro area	(1992-94)	Burnett et al., 1997b	all ages	Resp:	PM10	2.12 (0.90, 3.42)	28

*PM10<50 μg/m^3

CV=Cardiovascular, IHD=Ischemic Heart Disease

HF=Heart Failure, Dys=Dysrhythmia

Resp=respiratory, ED=Emergency Dept

COPD=Chronic Obstructive Pulmonary Disorder

Pneu=Pneumonia

City	Time Period	Author(s), Year Published	Age Group	Endpoint	Particle Type	% Increase per 10 µg/m³ and 95% CI	Mean (Bold=Median)
Montreal	Summers (1992-93)	Delfino et al., 1997a	under 65	Resp:ED	PM10	7.32 (2.00, 12.64)	22
Buffalo	May (1988)-Oct (1990)	Gwynn et al., 2000	all ages	CV:	PM10	1.14 (-0.66, 3.10)	24
Buffalo	May (1988)-Oct (1990)	Gwynn et al., 2000	all ages	Resp:	PM10	2.20 (0.80, 3.60)	24
London	(1992-94)	Hajat et al., 2001	0-14	Resp:Dr. visits for allergic rhinitis	PM10	5.67 (2.21, 9.45)	29
London	(1992-94)	Hajat et al., 2001	15-64	Resp:Dr. visits for allergic rhinitis	PM10	6.85 (4.59, 8.66)	29
Los Angeles, CA	(1992-95)	Linn et al., 2000	30 and older	CV:	PM10	0.65 (0.41, 0.89)	45
Los Angeles, CA	(1992-95)	Linn et al., 2000	30 and older	Resp:	PM10	0.66 (0.34, 1.00)	46
Detroit, MI	(1992-94)	Lippmann et al., 2000	65 and older	CV:Dys	CP	0.08 (-4.88, 5.76)	31
Detroit, MI	(1992-94)	Lippmann et al., 2000	65 and older	CV:HF	CP	2.08 (-1.32, 5.80)	31
Detroit, MI	(1992-94)	Lippmann et al., 2000	65 and older	CV:IHD	CP	4.20 (1.08, 7.56)	31

*PM10<50 µg/m³

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Resp=respiratory, ED=Emergency Dept

COPD=Chronic Obstructive Pulmonary Disorder

Pneu=Pneumonia

City	Time Period	Author(s), Year Published	Age Group	Endpoint	Particle Type	% Increase per 10mg/m^3 and 95% CI	Mean (Bold=Median)
Detroit, MI	(1992-94)	Lippmann et al., 2000	65 and older	Resp:COPD	CP	3.72 (-1.76, 10.00)	12
Detroit, MI	(1992-94)	Lippmann et al., 2000	65 and older	Resp:Pneu	CP	4.80 (0.32, 9.60)	12
Detroit, MI	(1992-94)	Lippmann et al., 2000	65 and older	CV:Dys	FP	1.28 (-2.60, 5.60)	31
Detroit, MI	(1992-94)	Lippmann et al., 2000	65 and older	CV:HF	FP	3.64(0.96, 6.48)	31
Detroit, MI	(1992-94)	Lippmann et al., 2000	65 and older	CV:IHD	FP	1.72 (-0.56, 4.16)	31
Detroit, MI	(1992-94)	Lippmann et al., 2000	65 and older	CV:Stroke	FP	0.72 (-2.12, 3.76)	31
Detroit, MI	(1992-94)	Lippmann et al., 2000	65 and older	Resp:COPD	FP	2.20 (-1.88, 6.80)	18
Detroit, MI	(1992-94)	Lippmann et al., 2000	65 and older	Resp:Pneu	FP	5.20 (1.48, 8.80)	18
Detroit, MI	(1992-94)	Lippmann et al., 2000	65 and older	CV:Dys	PM10	0.58 (-1.36, 2.74)	31
Detroit, MI	(1992-94)	Lippmann et al., 2000	65 and older	CV:HF	PM10	1.94 (0.04, 4.02)	31
Detroit, MI	(1992-94)	Lippmann et al., 2000	65 and older	CV:IHD	PM10	1.78 (0.10, 3.60)	31
Detroit, MI	(1992-94)	Lippmann et al., 2000	65 and older	CV:Stroke	PM10	0.96 (-1.10, 3.24)	31

*PM10<50 μg/m^3

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City	Time Period	Author(s), Year Published	Age Group	Endpoint	Particle Type	% Increase per 10mg/m^3 and 95% CI	Mean (Bold=Median)
Detroit, MI	(1992-94)	Lippmann et al., 2000	65 and older	Resp:Pneu	PM10	4.40 (1.66, 7.20)	31
Santa Clara Co, CA	Winters @20^{o}F (1988-92)	Lipsett et al., 1997	all ages	Resp:Asthma Emergency Dept Visits	PM10	6.94 (3.20, 11.30)	61
Paris	(1991-95)	Medina et al., 1997	all ages	Resp:Asthma House Visits	PM13	2.54 (0.82, 4.38)	25
Paris	(1991-95)	Medina et al., 1997	0-14	Resp:Asthma House Visits	PM13	8.30 (4.00, 13.36)	25
Paris	(1991-95)	Medina et al., 1997	15-64	Resp:Asthma House Visits	PM13	1.26 (-0.92, 3.70)	25
Los Angeles Co, CA	(1987-95)	Moolgavkar, 2000b	65 and older	CV:	FP	1.72 (1.00, 2.44)	**44**
Los Angeles Co, CA	(1987-95)	Moolgavkar, 2000b	20-64	CV:	FP	1.40 (0.72, 2.12)	**44**
Cook Co, IL	(1987-95)	Moolgavkar, 2000b	65 and older	CV:	PM10	0.84 (0.60, 1.10)	**35**
Los Angeles Co, CA	(1987-95)	Moolgavkar, 2000b	65 and older	CV:	PM10	0.64 (0.24, 1.06)	**44**
Los Angeles Co, CA	(1987-95)	Moolgavkar, 2000b	20-64	CV:	PM10	0.88 (0.44, 1.34)	**44**
Maricopa Co, AZ	(1987-95)	Moolgavkar, 2000b	65 and older	CV:	PM10	-0.48 (-1.38, 0.46)	**41**
Los Angeles Co, CA	(1987-95)	Moolgavkar, 2000c	0-19	Resp:COPD	CP	6.84 (3.56, 10.32)	**44**

*PM10<50 μg/m^3

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City	Time Period	Author(s), Year Published	Age Group	Endpoint	Particle Type	% Increase per 10mg/m^3 and 95% CI	Mean (Bold=Median)
Los Angeles Co, CA	(1987-95)	Moolgavkar, 2000c	65 and older	Resp:COPD	CP	2.04(-0.16, 4.36)	**44**
Los Angeles Co, CA	(1987-95)	Moolgavkar, 2000c	0-19	Resp:COPD	FP	1.72 (-0.04, 3.56)	**22**
Los Angeles Co, CA	(1987-95)	Moolgavkar, 2000c	20-64	Resp:COPD	FP	2.24 (0.76, 3.76)	**22**
Los Angeles Co, CA	(1987-95)	Moolgavkar, 2000c	65 and older	Resp:COPD	FP	2.04 (0.36, 3.76)	**22**
Los Angeles Co, CA	(1987-95)	Moolgavkar, 2000c	0-19	Resp:COPD	PM10	2.14 (0.88, 3.46)	**44**
Los Angeles Co, CA	(1987-95)	Moolgavkar, 2000c	20-64	Resp:COPD	PM10	1.30 (0.34, 2.30)	**44**
Los Angeles Co, CA	(1987-95)	Moolgavkar, 2000c	65 and older	Resp:COPD	PM10	1.22 (0.22, 2.26)	**44**
Minneapolis/St. Paul, MN	(1986-91)	Moolgavkar et al., 1997	65 and older	Resp:COPD+Pneu	PM10	1.74 (0.92, 2.60)	34
Birmingham, AL	(1986-91)	Moolgavkar et al., 1997	65 and older	Resp:COPD+Pneu	PM10	0.30 (-0.30, 0.92)	43
King County, WA	(1987-95)	Moolgavkar et al., 2000	all ages	Resp:COPD	FP	2.56 (0.36, 4.84)	30
King County, WA	(1987-95)	Moolgavkar et al., 2000	all ages	Resp:COPD	PM10	1.02 (0.00, 2.08)	30
Sydney	(1990-94)	Morgan et al., 1998	all ages	CV:	FP (bscat)	1.56 (0.44, 2.72)	10

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City	Time Period	Author(s), Year Published	Age Group	Endpoint	Particle Type	% Increase per 10 $\mu g/m^3$ and 95% CI	Mean (Bold=Median)
Los Angeles, CA	Wet seasons (1991-94)	Nauenberg and Basu, 1999	all ages	Resp:Asthma	PM10	3.24 (0.40, 6.00)	45
Spokane, WA	(1995-97)	Norris et al., 2000	under 65	Resp:Asthma Emergency Dept Visits	PM10	0.48 (-2.18, 3.52)	28
Seattle, WA	(1995-96)	Norris et al., 2000	under 18	Resp:Asthma Emergency Dept Visits	PM10	11.24 (-2.18, 3.52)	28
Santiago	(1992-93)	Ostro et al., 1999b	under 2	Resp:Lower Resp Clinic Visits	PM10	0.50 (0.04, 0.96)	109
Santiago	(1992-93)	Ostro et al., 1999b	2-14	Resp:Lower Resp Clinic Visits	PM10	0.74 (0.16, 1.34)	109
Edinburgh	(1992-95)	Prescott et al., 1998	under 65	CV:	PM10	0.40 (-2.50, 3.80)	21
Edinburgh	(1992-95)	Prescott et al., 1998	65 and older	CV:	PM10	2.48 (0.92, 4.18)	21
14 cities, US	(1985-94) range varies by city	Samet et al., 2000a	65 and older	CV:	PM10	1.10 (0.94, 1.24)	33
14 cities, US	(1985-94) range varies by city	Samet et al., 2000a	65 and older	Resp:COPD	PM10	1.50 (1.06, 1.96)	33

*PM10<50 $\mu g/m^3$

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Pneu=Pneumonia

City	Time Period	Author(s), Year Published	Age Group	Endpoint	Particle Type	% Increase per 10 $\mu g/m^3$ and 95% CI	Mean (Bold=Median)
14 cities, US	(1985-94) range varies by city	Samet et al., 2000a	65 and older	CV:	PM10*	1.52(1.20, 1.82)	33
Minneapolis/St. Paul, MN	(1986-89)	Schwartz, 1994c	65 and older	Resp:Pneu	PM10	0.12 (0.10, 0.13)	36
Minneapolis/St. Paul, MN	(1986-89)	Schwartz, 1994c	65 and older	Resp:COPD	PM10	0.16 (0.12, 0.21)	36
Tacoma, WA		Schwartz, 1995	65 and older	Resp:	PM10	2.00 (3.45, 0.64)	37
Cleveland, OH	(1988-90)	Schwartz, 1996	65 and older	Resp:	PM10	1.16 (0.10, 2.28)	43
Tuscon, AZ	(1988-90)	Schwartz, 1997	over 65	CV:	PM10	1.21 (0.22, 0.25)	42
8 cities, US	(1988-90)	Schwartz et al., 1999	65 and older	CV:	PM10	1.00 (0.74, 1.28)	**23-37**
Seattle, WA	(1987-94)	Shepard et al., 1999	under 65	Resp:Asthma	CP	4.44 (1.12, 8.04)	16
Seattle, WA	(1987-94)	Shepard et al., 1999	under 65	Resp:Asthma	FP	3.48 (1.32, 5.72)	17
Seattle, WA	(1987-94)	Shepard et al., 1999	under 65	Resp:Asthma	PM10	2.74 (1.10, 4.52)	32
Saint John	(1992-96)	Stieb et al., 2000	all ages	CV:	FP	6.04 (-0.12, 13.12)	9

*PM10<50 $\mu g/m^3$

CV=Cardiovascular, IHD=Ischemic Heart Disease

HF=Heart Failure, Dys=Dysrhythmia

Resp=respiratory, ED=Emergency Dept

COPD=Chronic Obstructive Pulmonary Disorder

Pneu=Pneumonia

City	Time Period	Author(s), Year Published	Age Group	Endpoint	Particle Type	% Increase per 10mg/m^3 and 95% CI	Mean (Bold=Median)
Atlanta, GA	Summers (1992-94)	Tolbert et al., 2000	under 17	Resp:Asthma Emergency Dept Visits	PM10	2.64 (0.24, 5.34)	39
Hong Kong	(1994-95)	Wong et al., 1999	all ages	CV:	PM10	0.60 (0.16, 1.08)	**45**

*PM10<50 μg/m^3

CV=Cardiovascular, IHD=Ischemic Heart Disease

HF=Heart Failure, Dys=Dysrhythmia

Resp=respiratory, ED=Emergency Dept

COPD=Chronic Obstructive Pulmonary Disorder

Pneu=Pneumonia

Table 7.4. Summary of Studies of Short-term Exposure and Respiratory Morbidity

City/Region	Time Period	Author(s), Year Published	Age Group, Additional Demographics	Particle Type	General Results	Mean (Bold=Median)
Rural and Urban areas, Holland	Winters (1992/93-94/95)	Boezen et al., 1999	7-11, N=632	PM10	Association with lower respiratory symptoms among children with both bronchial hyperresponsiveness and high total serum IgE.	Urban: 55, 42, and 31. Rural : 45, 44, 27
Alpine (rural southern CA)	Aug - Oct (1995)	Delfino et al., 1998	9-17, N=24, asthmatics	PM10	Association with asthma symptoms, especially children less frequently symptomatic for asthma on anti-inflammatory medication	31
Alpine (rural southern CA)	Summer (1994)	Delfino et al., 1997b	9-46, N=22, asthmatics	PM10	Association with inhaler use	26
Amsterdam, Holland	Early Summer (1995)	Gielen et al., 1997	7-13, N=61, majority asthmatics	PM10	Association with acute respiratory symptoms	31
Leiden University Medical Center, Holland	Summer (1995)	Hiltermann et al., 1998	18-55, N=60	PM10	Association with shortness of breath and bronchodilator use	40

City/Region	Time Period	Author(s), Year Published	Age Group, Additional Demographics	Particle Type	General Results	Mean (Bold=Median)
Reanalysis of several studies including Utah Valley, UT; Bennekom, Holland; Uniontown, PA; State College, PA	varies by study	Hoek et al., 1998	children	PM10	Significant decreases in PEF	varies by study
Los Angeles, CA	Summer (1992)	Ostro et al., 1995	7-12, N=83, african-american, asthmatics	PM10	Association with shortness of breath, particulary moderate and severe asthmatics.	56
Los Angeles, CA	Aug - Oct (1993)	Ostro et al., 2001	8-13, N=138, african-american asthmatics	PM10, FP	Association with new episodes of cough and extra asthma medication.	PM10=52, FP=41
Ausburg, Germany	Oct (1994)-June (1995); severe episode Jan 7-19 (1985)	Peters et al., 1997	25-64, N=3256	TSP	Association with increased plasma viscosity in both men and women when comparing severe pollution episode to the remainder of study.	TSP=47; severe episode TSP=98

City/Region	Time Period	Author(s), Year Published	Age Group, Additional Demographics	Particle Type	General Results	Mean (Bold=Median)
Utah Valley, UT	Winter (1990/91)	Pope and Dockery, 1992	10-12, N=79, split between those asymptomatic for asthma and those symptomatic for asthma but not on medications	PM10	Particularly symptomatic children, associations with respiratory symptoms and significant association with small decreases in PEF.	76
Mexico City	Apr (1991)-Feb (1992); 2 months	Romieu et al., 1996	5-13, N=71, mild asthmatics	PM10	Association with increased lower respiratory illness and decreased PEF.	167
New Haven, CT and Tacoma, WA	1988-1990	Schwartz et al., 1994	65 and older, all hospital admissions	PM10	Association with respiratory hospital admissions	41-New Haven; 37-Tacoma
Reanalysis of Harvard Six City Study, Uniontown and State College, PA	varies by study	Schwartz and Neas, 2000	children grades 2-5,	FP and CP	Association with lower respiratory symptoms; stronger effect with FP. Association with decreased PEF for FP.	varies by study
Kuopio, Finland	Spring (1995); six weeks	Tiittanen et al., 1999	8-13, N=49, children with chronic respiratory disease	PM10, FP, CP	Association with morning PEF and cough; strongest association for FP and CP.	PM10=28, FP=15

City/Region	Time Period	Author(s), Year Published	Age Group, Additional Demographics	Particle Type	General Results	Mean (Bold=Median)
urban and nonurban areas, Holland	Winters (1992/93-94/95)	van der Zee et al., 1999	7-11, N=795	PM10	Significant association with decreases in PEF and lower repsiratory symptoms in symptomatic children.	ranged 24-53
Port Alberni, British Columbia, Canada	May (1990)-Mar (1992)	Vedal et al., 1998	6-13, N=206 including 75 asthmatics	PM10	Associations with cough and phlegm and decreased PEF, particularly among asthmatics.	**22**
Seattle, WA	Nov (1993)-Aug (1995); 28-112 days	Yu et al., 2000	5-13, N=133, mild/moderate asthmatics	PM10, PM1.0 (nephelometry)	Association with asthma symptoms; strong association for PM1.0	PM10=25, PM1.0=10
Vinton, VA	Summer (1995)	Zhang et al., 2000	adult, N=673, mothers	PM10, FP, CP	Association with new episodes of rhinitis for CP.	??

1

Table 7.5. Relative Risk Estimates for Mortality Related to Average Annual PM2.5 -- Effect Modification by Education in Two Prospective Cohort Mortality Studies*

Study (Δ PM2.5)**	**Less than High School Education**	**High School Graduates**	**Post-high school education**
ACS (24.5)	1.27 (1.13 – 1.42)	1.20 (1.08 – 1.33)	1.05 (0.96 – 1.23)
Six-cities (18.6)	1.45 (1.13 – 1.85)	1.30 (0.98 – 1.73)	0.98 (0.72 – 1.36)

* - Pope et al. (1995); Dockery et al. (1993)

** - Δ PM2.5 = inter-quartile range of PM2.5 (annual average)

Table 7.6. Pollution-related mortality versus all-cause mortality in the elderly population.

City	First Author	% Share of Total Mortality for Elderly	% Share of Pollution-related Mortality for Elderly
Santiago, CH	Ostro (1996)	65	79
Mexico City	Loomis (1966)	57	68
London	Bremner (1999)	82	62
Bangkok	Ostro (1999)	66	73
Brisbane, AU	Simpson (1997)	81	90
Philadelphia	Kelsall (1997)	41	33

Table 7.7. Annual Health Benefits in California Associated with PM2.5 Annual Average of 12 μg/m^3.

Health Endpoint	Reference	% change per 10 μg/m^3 (± 95% CI)	Expected Incidence (cases/year)		
			Low	Mean	High
Mortality					
Long-Term Exposures Mortality (Age 30+)	Krewski et al., 2000	4.62 (1.20)	3,200	6,500	9,700
Short-Term Exposures Mortality (all ages)#	Schwartz et al. (1996)	1.43 (0.13)	2,100	2,600	3,100
Hospitalization					
COPD (ICD codes 490-492, 494-496), Age 65+	Samet et al., 2000	2.88 (1.39)	30	600	1,200
Pneumonia (ICD codes 480-487), Age 65+	Samet et al., 2000	2.07 (0.58)	400	900	1,300
Cardiovascular (ICD codes 390-429), Age 65+	Samet et al., 2000	1.19 (0.11)	1,300	1,500	1,800
Asthma (ICD codes 493), Age ≤ 64	Sheppard et al., 1999	2.57 (0.77)	90	500	800
Alternative Estimates of Hospitalization					
Circulatory (ICD codes 410-459), All ages	Van Den Eeden et al., 1999	0.80 (0.39)	90	2,100	4,200
Chronic respiratory (ICD codes 490-496), All ages	Van Den Eeden et al., 1999	2.70 (0.77)	600	1,500	2,300
Acute respiratory (ICD codes 460-519), All ages	Van Den Eeden et al., 1999	1.24 (0.47)	200	700	1,200
Minor Illness					
Lower respiratory symptoms, Age 7-14	Schwartz et al., 1994	18.23 (5.86)	81,000	209,000	323,000

Should not be added to estimate of long-term mortality, which includes some of the short-term effects.

Table 7.8. Annual Health Benefits in California Associated with PM10 Annual Average of 20 μg/m^3.

Health Endpoint	Reference	% change per 10 μg/m^3 (± 95% CI)	Expected Incidence (cases/year)		
			Low	Mean	High
Mortality					
Long-Term Exposures Mortality (Age 30+)*	Krewski et al., 2000	4.62 (1.20)	3,200	6,500	9,800
Short-Term Exposures Mortality (all Ages)#	Pooled California Studies (Chestnut & Mills, 2001)	0.838 (0.20)	1,600	3,000	4,600
Hospitalization					
COPD (ICD codes 490-492, 494-496), Age 65+	Samet et al., 2000	2.88 (1.39)	70	1,200	2,300
Pneumonia (ICD codes 480-487), Age 65+	Samet et al., 2000	2.07 (0.58)	800	1,700	2,600
Cardiovascular (ICD codes 390-429), Age 65+	Samet et al., 2000	1.19 (0.11)	2,500	3,100	3,600
Asthma (ICD codes 493), Age ≤ 64	Sheppard et al., 1999	2.57 (0.77)	200	800	1,500
Alternative Estimate of Hospitalization					
Circulatory (ICD codes 410-459), All ages	Van Den Eeden et al., 1999	0.80 (0.39)	200	4,300	8,300
Chronic respiratory (ICD codes 490-496), All ages	Van Den Eeden et al., 1999	2.70 (0.77)	1,300	2,900	4,500
Acute respiratory (ICD codes 460-519), All ages	Van Den Eeden et al., 1999	1.24 (0.47)	400	1,400	2,300
Minor Illness					
Lower respiratory symptoms, Age 7-14	Schwartz et al., 1994	18.23 (5.86)	161,000	389,000	573,000

Coefficient was multiplied by 0.5 assuming only the PM2.5 fraction of PM10 was associated with exposure.

Should not be added to estimate of long-term mortality, which includes some of the short-term effects.

8 Welfare Effects of Particulate Matter

8.1 Standards and "Welfare Effects"

"Welfare effects" includes all air pollutant impacts unrelated to human health. The manner in which these effects are evaluated depends on the legal authority for standard setting and how these effects bear on the standard in question. The California State standard setting environment is distinct from that under Federal law.

Under the Federal Clean Air Act (FCAA) (42 USC Ss 108 & 109) the National Ambient Air Quality Standard (NAAQS) for a particular pollutant consists of a "primary" standard aimed at protecting public health, and a "secondary" standard addressing welfare effects (if such effects exist). For gaseous air pollutants, such as ozone, the "primary-secondary" model allows the regulatory process to distinguish between an exposure (a specific concentration and duration) that causes human health impacts and other exposures that cause environmental and/or economic impacts.

Unlike a chemically homogeneous gaseous pollutant, particulate matter is a complex mixture of chemicals distributed over a wide range of particle sizes, with wide variation of chemical composition across particle size ranges. Moreover particle size and composition vary over time and between geographic areas. Consequently, the effects of particulate matter reflect its heterogeneous nature – different materials in different size ranges may have very different effects.

California law allows broad flexibility for air quality standards to address "public health, safety, and welfare, including, but not limited to, health, illness, irritation to the senses, aesthetic value, interference with visibility, and effects on the economy" [H&SC 39606(a)(2)]. In establishing the State PM10 Standard, the Air Resources Board declared that PM10 is "the fraction of inhalable particles which cause adverse health effects" and it should be "specifically addressed in a health-based standard" (ARB, 1982).

California has legal authority to define additional standards to specifically address other particulate matter effects. The PM10 standard is, therefore, not burdened with the requirement to cover all aspects of particulate matter pollution, and a separate State standard for "Visibility Reducing Particles" was created to address the dominant welfare effect of particulate matter - haze.

This section presents a brief overview of welfare effects and their regulation under State and Federal law to place the present PM10 review in the larger context of the role of particulate matter in the global environment.

8.2 Optical Effects: Visibility and Climate

The effects of particulate matter (aerosols) on visibility and climate are caused by the same optical processes. Visibility is reduced when aerosols interfere with light passing between an observer and a distant target; climate effects occur when aerosols interfere with incoming solar radiation or outgoing terrestrial radiation, changing the net energy balance between Sun and Earth. Where, how, and how intensely these interactions occur determines whether or not they are matters of regulatory concern. (The following discussion is highly simplified, the reader is referred to Friedlander (1977) for a full review of aerosol optics.)

8.2.1 How Particles Interact with Light

When a beam of electromagnetic radiation ("light") encounters the gases, particles and droplets that comprise the atmosphere, some light is scattered, some is absorbed, and some continues along its original path. The obscuring quality of a particular volume of air is termed "turbidity"; the experience of turbidity is the perception of "haze." The reduction of intensity of a beam of light as it moves through the atmosphere is termed "extinction," expressed as the "extinction coefficient" – the natural logarithm of the fractional change in intensity per unit distance (Middleton, 1952). Extinction is conventionally reported in units of "inverse megameters" (1/1,000,000m, or "Mm^{-1}"). Extinction is defined by the fundamental radiation transfer equation:

$$I_1 = I_0 \, e^{-Bext * d} \qquad (8.1)$$

where I_0 is the intensity of a beam at the beginning of the beam path, I_1 is the intensity at the end of the path, e is the root of natural logarithms (2.718…), B_{ext} is the extinction coefficient per unit distance, and d is the path length.

Under typical ambient conditions, extinction by various materials and processes is additive. Total extinction is the sum of scattering and absorption:

$$B_{ext} = B_{scat} + B_{abs}$$

Total extinction can be directly measured by observing the reduced intensity of a beam of light over a fixed distance, or scattering and absorption can be measured independently (monitoring methods are addressed below).

The strength of extinction is a function of the wavelength of the light, the density of the air, and the concentration, size and chemical composition of particles and droplets (aerosols). The extinction coefficient is additive, consisting of the sum of independent extinction due to n components of the atmosphere:

$$B_{ext} = \Sigma_{(1-n)} B_i * C \qquad (8.2)$$

where B_i is the extinction coefficient per unit mass for the i-th component, and C_i is the mass concentration of the i-th component.

8.2.2 Components of Extinction

Extinction can be represented as the linear sum of four generic components: scattering and absorption by both gases and particles. This is represented by the equation:

$$B_{ext} = B_{sg} + B_{ag} + B_{sp} + B_{ap}$$

Assessing the causes of strong extinction usually involves addressing each of these components separately.

Under typical ambient air conditions, B_{sg}, also known as "Rayleigh scattering," is a function of air density (thus a function of altitude), and proportional to the fourth power of wave number (inverse of wavelength):

$$B_{sg} (Mm^{-1}) = 12 * \rho \quad (500/\lambda)^4 / (.00123)$$

where ρ = air density (g/cm^3) and λ = wavelength (nm).

For green light (the middle of the visible range) at standard conditions:

$$B_{sp} = 12 \; Mm^{-1}$$

Light absorption by gases, B_{ag}, for clean air and visible light, is practically zero. In urban atmospheres, nitrogen dioxide absorbs blue light, causing a yellowing of the sky and

distant targets. Outside the visible range, gaseous components of the atmosphere exhibit strong absorption at various wavelengths in both the ultraviolet (especially ozone) and infrared (especially water vapor and carbon dioxide) portions of the electromagnetic spectrum.

B_{sp} is generally the largest component of extinction. For a given mass of aerosol, the largest determinant of scattering is particle size. If the aerosol size distribution and composition are held constant, then, for typical atmospheric particle loads, scattering will be proportional to particle concentration. If the particles contain hydrophilic chemicals (e.g., nitrates, sulfates), the size distribution may change with humidity. Raising humidity will promote particle growth through absorption of water, increasing scattering with no change in pollutant concentration.

The scattering power for a particular amount of aerosol can be expressed as effective surface area - the scattering cross-section (cm^2). If the volume or mass or aerosol is known, "scattering efficiency" can be expressed, respectively, as the volumetric scattering efficiency (cm^2/cm^3) or the mass scattering efficiency (cm^2/g). Using appropriate units:

$$B_{sp}\ (Mm^{-1}) = \text{Efficiency}\ (m^2/g) * \text{Concentration}\ (\mu g/m^3)$$

(see eqn. 8.2 above)

Particle size is very important for scattering (Friedlander, 1977).

The relationship between size and particle scattering efficiency for monochromatic light (a single wavelength) is plotted in Figure 8.1. The scattering cross section of particles much smaller than the wavelength of the light being scattered ($d/\lambda<0.1$) is negligible. For particles much larger than the wavelength ($d/\lambda>10$), the effective cross section tends toward twice the actual cross section. For particles near the wavelength, complex electrical interaction between light waves and particles accentuates scattering, increasing it to about 4 times the particle cross section for particles near $d/\lambda = 2$ (a process known as Mie scattering).

Expanding to all wavelengths of visible light, scattering efficiency is near zero for particles less than .05 μm diameter, less than 1 m^2/cm^3 for particles near 0.1 μm diameter, rises to a peak at about 10 m^2/cm3 in the range .4 to .7 μm diameter, then falls to less than 1 m^2/cm^3 for particles greater than 2 μm diameter and continues to decrease as the inverse of diameter for larger particles (Friedlander, 1977). Applying these physical characteristics to observed ambient aerosol size distributions, Friedlander (1977) calculated that light scattering is dominated by the population of particles between 0.2 and 2 μm diameter (Figure 8.1).

Absorption by aerosols, B_{ap}, is essentially a function of particle composition and total aerosol surface area. The strongest and most common absorbing aerosols are composed of nearly pure carbon ("elemental carbon", "EC", or "soot"); some soil materials also absorb some visible light (especially iron oxides), but with only a fraction of the efficiency of carbonaceous aerosols. Aerosols composed of a mixture of EC and other materials exhibit intermediate absorption efficiencies, roughly proportional to their EC content. Absorption is moderately sensitive to particle size. Very small particles ($d/\lambda<0.1$) don't interact efficiently with light waves. For particles with $d/\lambda>0.3$, absorption is roughly linear to total particle surface area, with the influence of particle size driven by the geometric decrease in surface area/volume ratio as size increases ($m^2/cm^3 \propto d^{2/3}$).

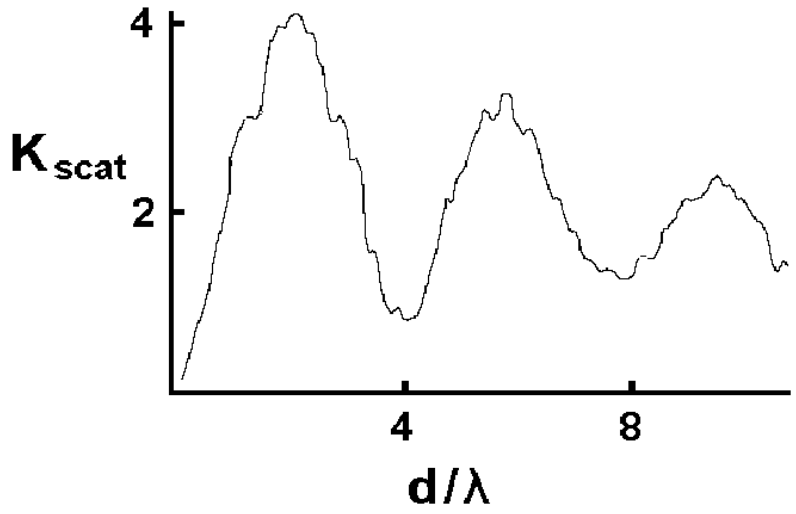

Figure 8.1a Monochromatic single particle scattering (Mie scattering; Friedlander, 1977).

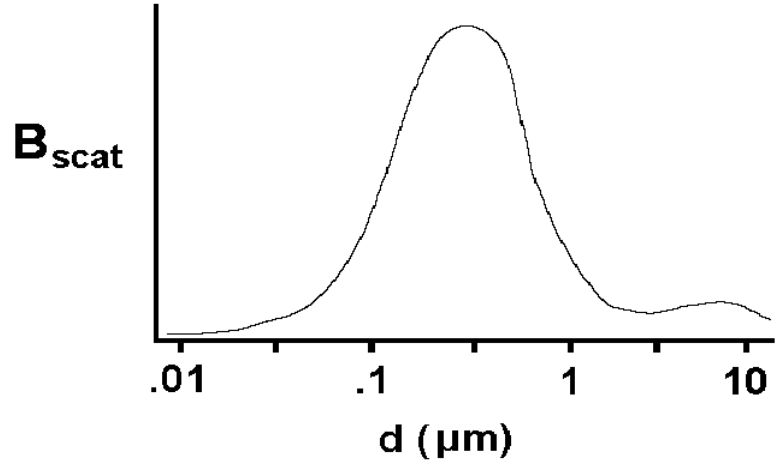

Figure 8.1b Particle light scattering vs. size in a typical urban aerosol (Friedlander, 1977).

8.2.3 PM10 and Extinction

Uniting the foregoing theoretical discussion and the discussion of PM composition in an earlier chapter (Chapter 3), it is evident that the smaller particles within PM10 (i.e., those below 2.5 μm diameter) play the dominant role in light extinction. Analysis of detailed aerosol data from over 5000 samples taken at 36 undeveloped rural sites across the United States (Sisler, et al., 1993) indicates that fine aerosols (PM2.5) exhibit 2 to 20 times more extinction efficiency per unit mass than do coarse particles (i.e., between 2.5 and 10 μm diameter), depending on chemical composition and relative humidity. For remote rural sites in California, their data (Sisler, et al., 1993) show that average coarse material light extinction is consistently less than 10 percent of the total. In urban areas, where fine material is more abundant, the coarse particle contribution to extinction is frequently even smaller.

8.2.4 Visibility

"Vision" is a psychophysical process involving light focusing and perception by the eye and image formation and interpretation by the brain. The process is subject to basic physiological limitations such as light sensitivity, spatial resolution, and color differentiation. Psychological processes control the brain's conversion of optic nerve signals into the perceptual components of vision, such as image formation, object recognition, and esthetic appreciation. Visual acuity varies among individuals due to interacting factors of physical and perceptual capabilities and acquired skill due to training and experience [this discussion is necessarily simplified, the reader is referred to Middleton (1952), ARB (1989), and Malm (1999)].

"Visibility" refers to the perceptibility of a distant target or scene. Variation of illumination, contrast, color, spatial frequency (target size and detail), background, foreground, etc., and the psychophysical variations among potential viewers combine to make "visibility" a very subjective concept. Managing visibility requires developing policy tools (such as air quality standards) that link physical qualities of the atmosphere to the subjective human experience of haze (ARB, 1989). This requires accepting a fundamental abstraction: regulating and managing the optical density of the air is a reasonable substitute for regulating the quality of human visual experience (ARB, 1989).

8.2.5 Measuring Visibility

8.2.5.1 Visual Range

In order to characterize atmospheric turbidity consistently and repeatably, measurements need to be standardized. "Visual Range" (V_r), in the parlance of meteorology or air pollution, is an operationally defined observation: the greatest distance at which a large black object can be distinguished from the background sky around a majority of the horizon circle. This method reduces the variation among definitions of "visibility," but imposes other strict limitations by requiring sites with clear views of the horizon in all directions and dark objects to view at varying distances in all directions. Moreover, it does not address differences among viewers. Nonetheless, visual range data are the best source of historical visibility information (Trijonis, 1980).

Visual Range data from many stations are significantly biased by lack of appropriate viewing targets. Historically, most Visual Range data have been recorded as part of routine weather observations at airports. Since low visibility impairs airport operations, "Airport Visibility" records are often biased toward reporting low visibility events, while

moderate and good visibility are frequently grouped together as “greater than 10 miles” or “greater than 30 miles” (Trijonis, 1980). As weather observations have been increasingly automated, “Meteorological Visibility” (Visual Range) observations at many locations have been replaced with instruments calibrated to replicate human observations; unfortunately, these instrumental records also replicate the bias toward measuring low visibility.

Visual Range from airport observations can be related to extinction if appropriate assumptions are applied. Human perception is much more sensitive to contrast than absolute light intensity (Middleton, 1952), so Visual Range can be restated as the distance at which a dark target (inherent contrast with the background sky $\cong$ 100%) is barely discernable to a human observer [‘liminal contrast” threshold for detection $\cong$ 5% (Trijonis, 1980)]. The fundamental radiation transfer equation (eq. 1) applies for contrast as well as for intensity, so, substituting contrast for intensity in eq. 1 gives:

$$C_1 = C_0 \, e^{-Bext * d} \qquad (8.3)$$

where C_0 is the scene contrast at the target and C_1 is the apparent scene contrast at the viewer’s location.

Algebraically transforming eq. 3 to relate distance (d, or in this case, V_r) to extinction (B_{ext}) and using the contrast assumptions above and units of Mm^{-1} gives:

$$V_r = 2996 / B_{ext} \qquad (8.4)$$

where V_r is in km,

or

$$V_r = 1857 / B_{ext}$$

where V_r is in miles.

The relationship in eq. 4 is generally known as the Koschmieder equation (Middleton, 1952).

Correcting for the limitations of airport data, Trijonis (1980) compiled a statewide assessment of visibility in California. Although there have been some reductions in aerosol loading in parts of the state, the general patterns he found still exist. Figure 8.2 shows Trijonis’ map of average visual range in California.

8.2.5.2 Instrumental Measurements

Since meteorological records are imperfect sources of visibility data, both Federal and California visibility monitoring programs use specialized monitoring methods designed to characterize “visual air quality” in a manner compatible with routine air quality management programs.

By measuring the physical property of “extinction” or its components (scattering and absorption) instrumentally, the “human factor” is eliminated altogether. Extinction can be related to measured aerosol characteristics (mass, size, chemical composition, etc.) both empirically (e.g., through regression analyses) and by calculating extinction “from first principles” using detailed knowledge of aerosol characteristics. These approaches allow management of visual air quality through the same types of measurement, modeling, and control programs that are used for other air quality purposes.

California’s instrumental measurement of extinction, California Method “V”, consists of side-by-side measurements of light scattering using a nephelometer and light absorption

on a filter (modified from the "Coefficient of Haze" protocol), and supported by measurements of relative humidity (RH) (ARB, 1989). This provides direct observation of aerosol optical properties at the location of the monitoring site.

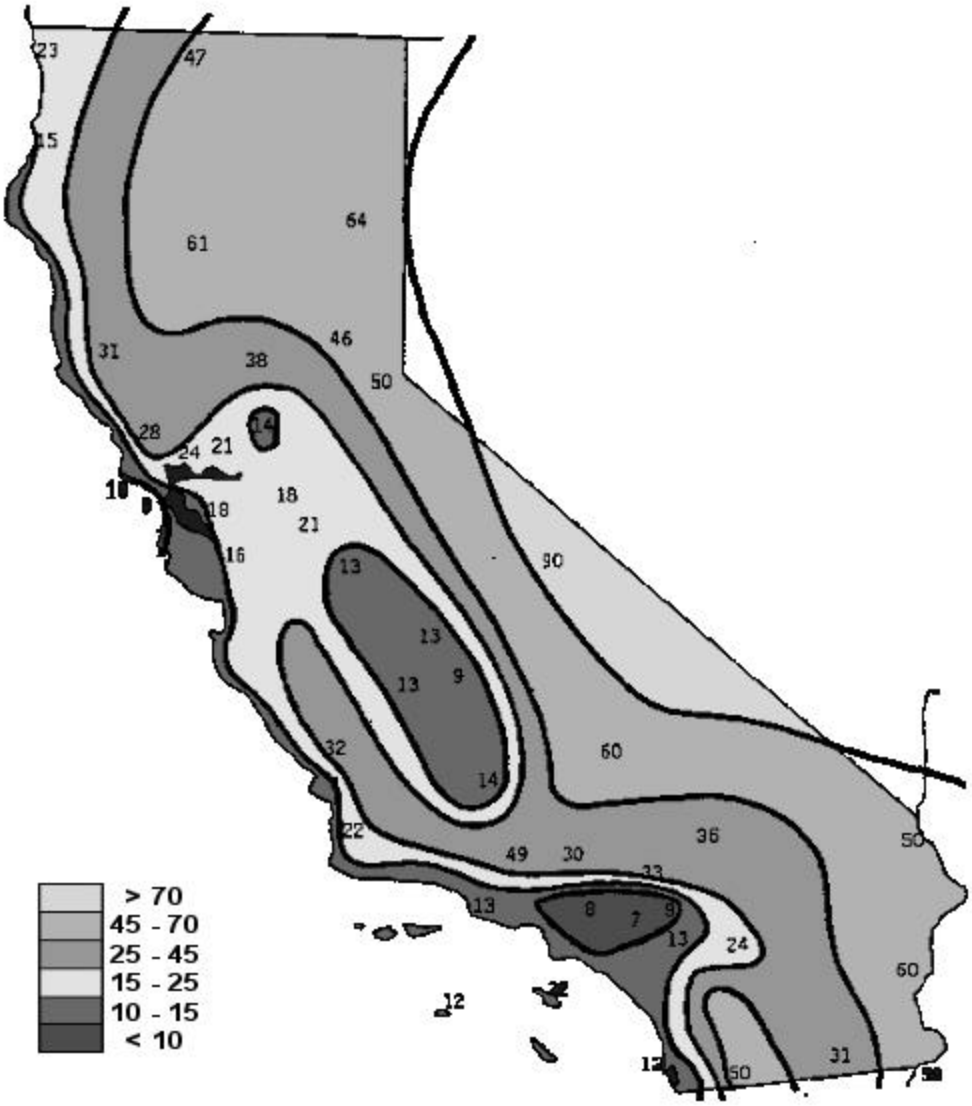

Figure 8.2 Average Annual Visual Range in California (Trijonis, 1980)

Determining the causes of the observed extinction depends on additional aerosol monitoring to identify the particular aerosol components present when visibility is poor, and then linking them to emission sources. Optical data (COH & nephelometer) consistent with Method V are available from 15 sites in the state. To date,

implementation of full Method V visibility monitoring (i.e., including RH) has been restricted to a few sites in the South Coast Air Basin and the Lake County Air Basin.

The United States national visibility monitoring is done by the Interagency Monitoring of Protected Visual Environments (IMPROVE) program (Sisler, et al., 1993). The primary IMPROVE protocol consists of size-selective aerosol collection (total PM10 mass and PM2.5 mass and elemental analysis) supported by a long-path transmissometer to measure total extinction over a fixed sight path near the monitoring site. Measurements of light absorption are taken from the PM2.5 particle filters; subtracting absorption from total extinction gives a measure of scattering. Some IMPROVE sites also employ nephelometers. Because the IMPROVE program combines optical and aerosol monitoring, the particular pollutants causing low visibility at an IMPROVE site can be assessed directly by analyzing the monitoring data. The IMPROVE network in California consists of six sites with records beginning in 1989, and 8 additional sites added in 2000.

8.3 Effects: Aesthetic, Economic, and Operational

Low visibility can result from natural causes (e.g., fog, volcanic eruption, forest fire smoke), from purely anthropogenic causes (e.g., industrial smoke, diesel exhaust), from mixtures of natural and anthropogenic aerosols (e.g., “agricultural haze” consisting of dust and combustion products), or from interactions of natural processes and anthropogenic activities (e.g., nitrate haze, smoke from prescribed fires). The fact of low visibility is not, of itself, cause for regulatory concern; rather, it is the combination of human cause and adverse effect on human activity that drives visibility regulation. While instrumental measurement and source identification can quantify the anthropogenic factors in the timing and intensity of poor visual air quality, identifying undesirable effects and determining appropriate levels of controls is wholly in the realm of policymaking.

There are three broad categories of effects due to reduced visibility: aesthetic, economic, and operational. Aesthetic effects, such as impairment of vistas in national parks, animate the present National visibility program. Economic effects, such as reduced tourism or depressed real estate values, are largely a secondary impact of aesthetic effects. Finally, operational impacts arise when low visibility interferes with “business as usual” such as airport operations, or causes short-term calamity (e.g., chain reaction accidents on freeways). Establishing visual air quality goals for each type of impairment involves balancing the effort and cost of control against the “value” (social and financial) of expected reductions in the frequency and intensity of visual impairment.

8.3.1 Aesthetic Effects

Aesthetic effects dominate the visibility regulatory landscape. The Federal visibility protection program derives from a tradition of National Park and Wilderness conservation based on eliminating all traces of human activity and preserving “pristine nature” in undisturbed enclaves – defined in the CAA as “Class I” areas. The FCAA defines the “national goal” as ‘‘prevention of any future, and the remedying of any existing, impairment of visibility in Class I areas which impairment results from manmade air pollution’’ (42 USC Sect. 169A).

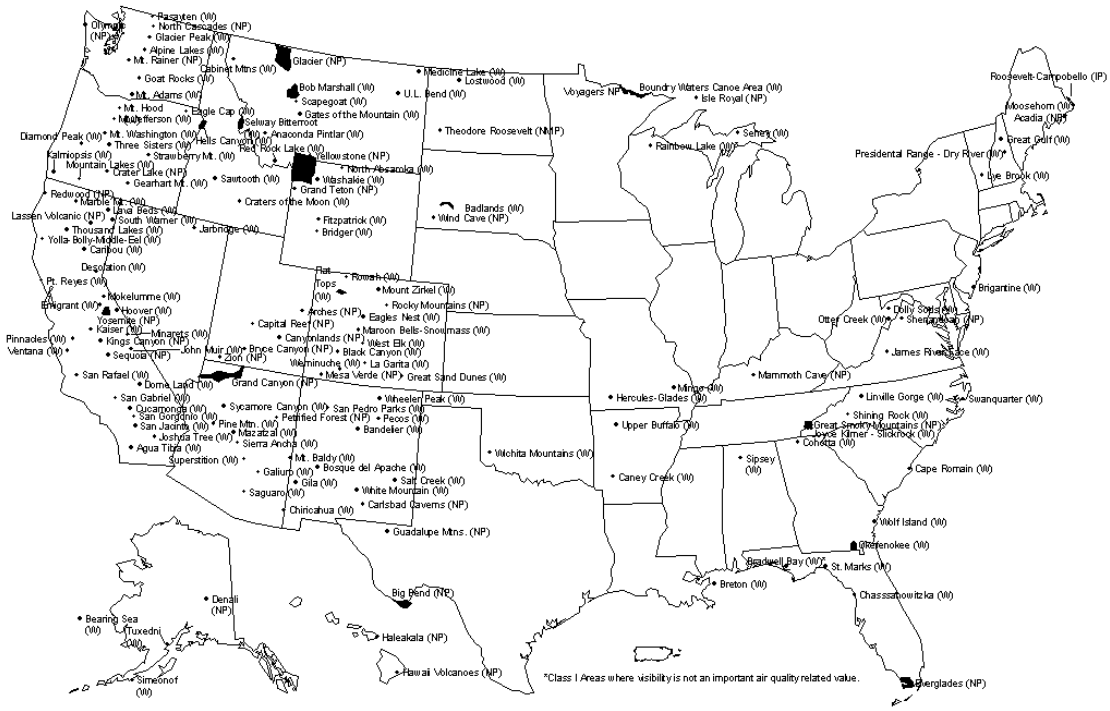

Figure 8.3 Class I Areas.

By contrast, California's State standard for Visibility Reducing Particles follows a pattern derived from health-based air quality regulation. California applies a single minimum visibility value (maximum extinction level) uniformly across an entire air basin (presently, the statewide standard applies in all air basins except Lake Tahoe, where the standard is much more stringent). The logic of using a single-value standard is that regulating emissions that cause low visibility events will necessarily limit the frequency and severity of all levels of impairment, thus regulating the human experience of intense haze will also reduce the experience of intermediate levels of haze (ARB, 1989). The level of the standard represents a policy judgement that identifies regionally appropriate visibility goals (hence a more stringent standard for the Lake Tahoe air basin than for other areas).

8.3.2 Economic Effects

The economic effects of reduced visibility appear in the form of reduced prices for real estate (especially "view" properties), reduced demand for visibility-related recreation, and diffuse effects of perceived degraded environmental quality. (Delucchi, et al., 1996; Trijonis et al., 1985, Rowe, and Chestnut, 1982). Measuring the economic value of visibility (or the cost of impairment) involves translating human preference into monetary value – known as "willingness to pay" (WTP).

There are two general approaches to measure WTP. Surveys asking respondents to set a value on a change in environmental quality are termed "stated preference" methods. Studies using statistical analysis of differential prices in real markets to infer the actual value of environmental amenities are termed "revealed preference" methods.

Loehman et al. (1994) measured the visibility and health risk WTP in the San Francisco Bay Area using stated preference data from a 1980 survey. Their methodology established three air quality classes (good, fair, poor; equivalent to V_r>10 mi., 10>V_r>6 mi., and V_r<6 mi.) and assigned respondents by residence to 5 sequentially ranked areas based on frequency and severity of pollution based on analysis of daily airport visibility data from around the region (we estimate equivalent PM10 cutpoint ranges as: V_r = 10 mi., 45-90 $\mu g/m^3$; V_r = 6 mi., 75-150 $\mu g/m^3$ depending on particle chemistry and size distributions). Respondents were asked to state how much they would pay per month to move up or to avoid moving down in air quality along the zonal scale. They found that overall individual WTP for visibility was about $0.10 per month (1980$) for each additional day per year of good air quality. It is interesting to note that this study also detected a "risk aversion" response. While visibility valuations were nearly symmetrical for improvement or avoidance, avoiding deterioration generally scored higher than improving air quality for health. Health based WTP to move up was relatively flat across all potential one-step changes, but WTP to avoid moving down increased with deteriorating air quality.

Trijonis et al. (1985) used the revealed preference method applied through multiple regression to analyze the value of visibility for residential real estate in California. Although somewhat dated, this study provides considerable insight into the effect of model formulation and variable specification on detecting WTP. Using a hybrid regression/principle component approach they eliminated the effects of spatial covariance between community characteristics and visibility, then tested various model forms for their explanatory power. Reporting the range of benefits calculated by the three best models for each area, they found, for a ten percent improvement in visibility, average home selling price in southern California would increase by 0.7 to 2.1 percent, while in the San Francisco Bay area, sales price would rise by 1.4 to 2.5 percent. Integrating over regional sales reported for 1978-79 produced economic benefits in the real estate sector of $250M to $617M (1979$) per year in southern California; and $190M to $220M (1979$) per year for the San Francisco Bay area. The breadth of analyses and use of multiple functional forms gives these results strong credibility and it is likely that they span the range of potential "true" values for visibility. There are no studies that address the current (2001) real estate market in California, but California's spatial patterns of both real estate values and visibility reduction are still much like they were in 1980, so it is reasonable to assume that similar percentage value increments apply to today's vastly more valuable real estate stock.

In the socioeconomic assessment of the Southern California Air Quality Management District's (SCAQMD) 1997 Air Quality Plan (Lieu, 1996), SCAQMD staff constructed estimates of the economic value of improved visibility derived from both the revealed and stated preference methods. They reported aggregate annual benefits of $109 million in 2000 and nearly $1.1 billion in 2010; resulting in average annual benefits over the period 1997-2010 of $473M.

8.3.2.1 Controlling Both PM10 and Visibility Reducing Particles

The economic studies and the SCAQMD valuation discussed here were based on either modest incremental changes in air quality or assessing the ancillary benefits

accompanying attainment of the annual 24-hour maximum health-based Federal PM10 standard (150 $\mu g/m^3$). The SCAQMD study assumes that all gains are achieved when the PM10 standard is attained. Although unreported in the literature, it is reasonable to expect that there would be additional benefits gained in attaining the State PM10 standard (at the time of the SCAQMD study the California standard was roughly 1/3 the level of the Federal standard) or the State Visibility Reducing Particles standard. While the reported data demonstrate that improving visibility has substantial economic benefits, it is difficult to interpret these findings in relation to other target extinction levels or to extrapolate these findings to other areas of California. A full evaluation of the statewide benefits of attaining alternative PM or visibility standards has yet to be done.

8.3.3 Operational Effects

Operational impacts of low visibility vary depending on the sensitivity of individual activities to visibility impairment.

8.3.3.1 Roadways

Motor vehicle traffic has a low-sensitivity to PM-caused visibility impairment. Highway traffic requires "good visibility" for safe vehicle flow, yet traffic is not very sensitive to particulate air pollution. Highway visibility is "good" when drivers can clearly see vehicles, objects, or intersections far enough ahead to react to traffic conditions and maintain safe distance from other vehicles. This generally requires sight distances in the range of tens to hundreds of meters (AASHTO, 2001). In dry weather, very high particle concentrations are required to create light extinction levels sufficient to impair vehicle traffic (e.g., a V_r of 500 m implies fine particle concentrations in the range from 1300 to 2500 $\mu g/m^3$). Such high particle concentrations are generally due to short term local sources such as excavation dust, fires, or "dust devils" – events typically not detected by routine monitoring and thus must be regulated by nuisance rules, rather than through air quality standards.

8.3.3.2 Airports

Airport operations, like road traffic, require "good" visibility, but the higher speeds and greater distances involved translate into greater sensitivity to particulate extinction. Ground operation minima are very short – comparable to those for highways [FAA requires airports to begin "low visibility operations" when visual range is less than 1200 ft. (0.74 km) (FAA, 1996)]. Safe flight operations require that pilots have the ability to see an airfield well enough to land, to avoid land-based obstacles or other aircraft, and to generally operate safely under Visual Flight Rules (VFR); for this the FAA has established minimum visibility (V_r) for unrestricted operations at 3 miles (5.1 km) (FAA, 1996). This translates to PM10 concentrations ranging from 130 to 250 $\mu g/m^3$, depending on aerosol conditions.

8.3.3.3 Aircraft Flight Testing

California is home to the two most heavily used flight test facilities in the United States. Air space over the eastern Sierra and the western Mojave desert is reserved for the joint use of Air Force, NASA, and Army testing operations based at Edwards Air Force Base in Antelope Valley and Navy test operations based at China Lake in Indian Wells Valley. These facilities were sited in this region because of their year-round flying weather, excellent visibility, and proximity to California's aerospace industry. Activities at these facilities directly employ over 10,000 people and are the mainstay of the western Mojave regional economy. Unlike typical aviation, these facilities are extremely sensitive to reduced visibility because they employ optical tracking and recording of flight tests using

powerful ground-based telescopic movie and video systems. Tracking each test from multiple sites, engineers are able to reconstruct flight dynamics of test or target aircraft, guided missiles, parachutes, or other test objects independent of onboard instrumentation (in some tests, onboard instrumentation is impossible, and the optical tracking is the sole flight record). To accomplish these tests, cameras must be able to track small objects in the sky from distances up to 20 miles (32 km) (VanCuren, 1982). In order to evaluate the threat to these operations due to air pollution, the Department of Defense (DoD) conducted the Research on Operations Limiting Visual Extinction (RESOLVE) project, an intensive visibility assessment in the region in the late 1980s (Trijonis, et al., 1988). While the DoD has not established absolute minimum visibility requirements for its operations, the RESOLVE study identified anthropogenic pollutants as episodically contributing to reduced operational capability, and DoD adopted a policy of working with local, State and Federal air quality regulators to prevent further degradation in the study area. Conditions deemed adverse in the RESOLVE context are associated with V_r below about 80 km (48 mi), or PM2.5 on the order of 10 $\mu g/m^3$ or greater.

8.3.4 Visibility Regulation

8.3.4.1 Federal Regional Haze Program

The FCAA defines a “national goal” of the ‘‘prevention of any future, and the remedying of any existing, impairment of visibility in Class I areas which impairment results from manmade air pollution’’ (42 USC Sect. 169A). The program has two parts, one addressing the impacts of individual large air pollution sources (“Reasonably Attributable Impairment”- RAI) and the other addressing the cumulative effects of all sources (“Regional Haze”).

The RAI program [40 CFR section 51.301(s)] is based on studying the direct aerosol impacts (termed “plume blight”) of large pollution sources or small groups of sources such as smelters or power plants, and requiring controls on new sources or retrofits on existing sources to reduce their impacts below the threshold of perceptibility. The best-known example of RAI is the case of the Navajo Generating Station at Page, AZ, which was ordered to install additional emission controls after it was found to impact Grand Canyon National Park. No such RAI pairing of a large source and a Class I area has been identified in California.

The Regional Haze program (EPA, 1999) is intended to address the cumulative, diffuse effects of all air pollution sources in a region. Regional Haze involves virtually all sources distributed over a large area (a state or multiple states) and effects on one or many Class I areas. The Regional Haze program does not establish a single visual air quality goal; rather it requires that each State must determine, on a case-by-case basis, “natural conditions” at each Class I area within its boundaries. “Natural” conditions must be represented as a range of visual air quality, and the national goal is interpreted as requiring that emissions be controlled to bring ambient conditions for the best 20% of days to approximate the best 20% of “natural” conditions, and that the worst 20% of days be indistinguishable from the worst 20% of “natural” conditions. The 156 Class I areas in the United States are mapped in Figure 8.3.

California’s responsibilities under the Regional Haze rules cover 29 in-State Class I areas, and an as yet undefined number of Class I sites in neighboring states.

Current visibility conditions at Class I areas in California range from near-pristine conditions at Redwood and Lassen Volcanic National Parks to substantially degraded at

Sequoia National Park and San Gorgonio Wilderness. Although specific goals have not yet been set for California Class I areas, the likely range of such goals can be inferred from data for the cleaner IMPROVE sites. PM10 at Redwood National Park (a "clean" low altitude coastal site) has a long-term mean around 12 $\mu g/m^3$ and rarely exceeds 30 $\mu g/m^3$. At Lassen Volcanic National Park (a "clean" montane site) long-term mean PM10 is below 10 $\mu g/m^3$ and rarely exceeds 20 $\mu g/m^3$.

8.3.4.2 California AAQS for Visibility Reducing Particles

The California State Ambient Air Quality Standard for Visibility Reducing Particles (VRP) represents a policy judgement that a certain minimum degree of visibility is conducive to public welfare, regardless of location. This policy is manifested as a Statewide minimum dry air particle extinction limit of 0.23/km (230 Mm^{-1}) averaged from 9 AM to 5 PM (PST) when Relative Humidity (RH) is less than 70 percent. This is roughly equivalent to V_r = 10 miles. The standard is 0.07/km (70 Mm^{-1}) for the Lake Tahoe Air Basin (roughly equivalent to V_r = 30 miles). Equivalent PM10 concentrations when this standard is just met range from about 50 $\mu g/m^3$ for a fine particle dominated urban setting (e.g., Sacramento in winter) to 90 or more $\mu g/m^3$ for a mixture of coarse and fine particles (e.g., Central Valley summer). The Lake Tahoe VRP limit equates to PM10 concentrations ranging from about 16 to 25 $\mu g/m^3$ over a similar range of aerosol characteristics.

State law permits the Board to adopt other standards for any Air Basin, although to date only Lake Tahoe has been singled out for additional protection.

8.4 Climate

Anthropogenic effects on climate have become very important international scientific and political issues. Understanding the scale of these effects, their causes, and anticipated harm, and identifying potential corrective actions are the subjects of major research programs. Beginning in the late 1980's, the World Meteorological Organization (WMO) and United Nations Environment Program (UNEP) have jointly sponsored the Intergovernmental Panel on Climate Change (IPCC), which has become the major international clearinghouse for assessing climate change (IPCC, 2001b) (the brief discussion presented here is largely based on the IPCC 2001 reports.) The initial focus of concern, both scientifically and for managing climate, was on so-called Green House Gases (GHGs) – CO_2, CH_4, etc. - but research over the last two decades has demonstrated that particles, too, have the potential to significantly alter climate processes.

Particles impact climate directly by modifying Earth's radiation balance through their interaction with both long wave (infrared) and short wave (visible) light, and indirectly by their role as condensation nuclei in cloud formation. This effect is termed "radiative forcing." Depending on chemistry, timing, and location, particles may either heat or cool the atmosphere.

Positive radiative forcing warms Earth's surface and lower atmosphere. Negative radiative forcing cools them. Natural factors, such as changes in solar output, explosive volcanic activity, snow, or cloud cover can also have radiative forcing effects. The planetary radiation balance is the net sum of all positive and negative forcing occurring together. Thus an effect such as climate warming by positive infrared forcing due to increasing CO_2 concentrations may be offset by negative forcing due to visible light scattering by "white" aerosols (e.g., sulfates) or enhanced by warming due to infrared and visible light absorption by "black" aerosols ("soot").

Determining the impact of anthropogenic PM emissions on climate requires properly accounting for all radiative forcing, natural and manmade, then determining the shift in net radiation that would occur if the anthropogenic component were removed, and finally calculating the change in climate that would result from that shift in radiation. While this is simple in concept, it is very difficult to implement because:

- We do not have a good inventory of all the aerosols in Earth's atmosphere.
- We do not know with certainty how much aerosol in Earth's atmosphere is due to anthropogenic activity.
- We do not know global PM emission and ambient aerosol distribution patterns with sufficient temporal and spatial resolution.
- We do not know how to partition secondary aerosol effects, such as cloud formation, between natural and anthropogenic condensation nuclei.
- We do not know how what co-effects would accrue to global-scale PM emission controls (CO_2 reduction, altered surface albedo, etc.).
- We do not have climate models with sufficient precision to reliably perform the climate effect calculation.

Figure 8.4 shows the relative positive or negative radiative forcing from various components of the climate system, with an assessment of the degree of certainty of climate knowledge in each area noted along the bottom of the figure. The major aerosol classes are briefly reviewed below.

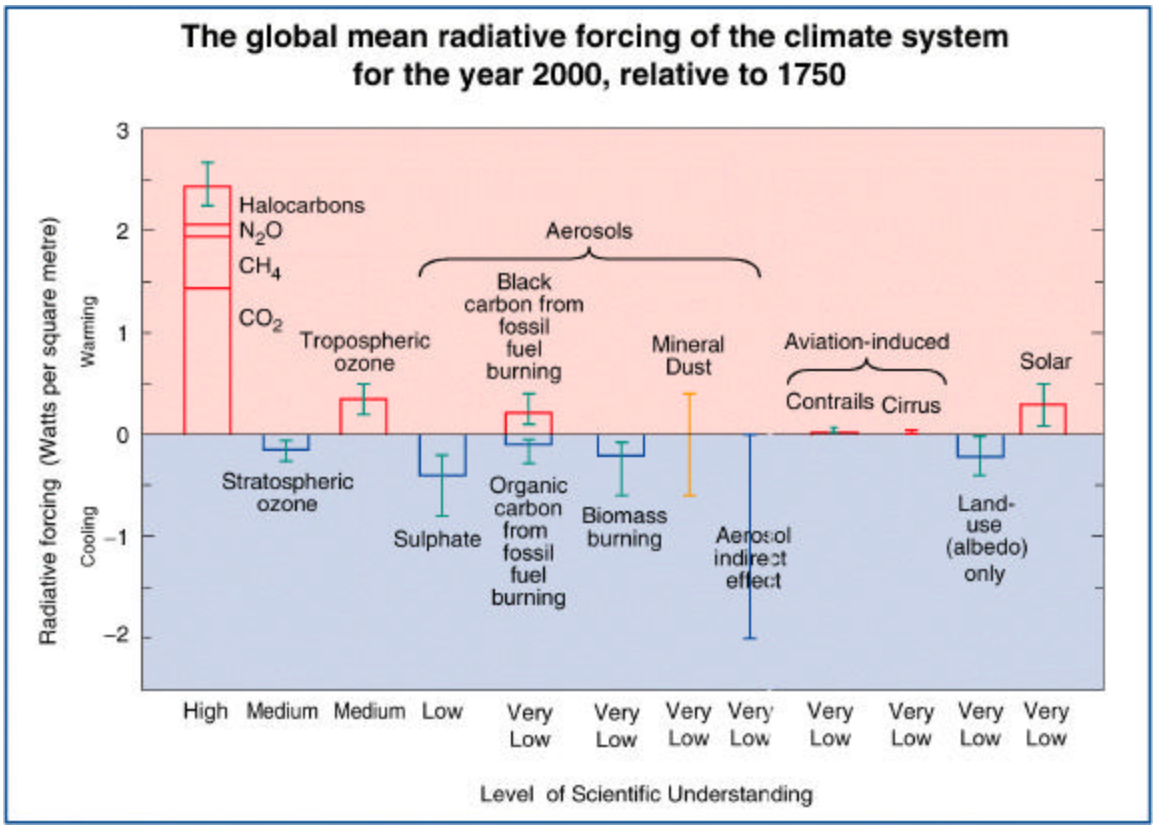

Figure 8.4 Summary of intensity and degree of scientific certainty of climate forcing by anthropogenic pollutants; note that aerosols' effects are both significant and highly uncertain (IPCC, 2001a).

8.4.1 Sulfate

The vast majority of sulfate aerosols are formed by the oxidation of gaseous sulfur compounds into sulfuric acid, which then combines with a metallic or alkaline ion to form a stable salt (Na_2SO_4, Mg_2SO_4, $(NH_4)_2SO_4$, etc.). Sulfate aerosols mostly form in heterogeneous (gas, droplet, and particle) atmospheric conversion, which tends to concentrate sulfate in fine aerosols (<2:m diameter). When both humidity and sulfuric acid concentrations are high and sufficient neutralizing ions are not present, a liquid phase sulfuric acid aerosol can form.

Due to the hygroscopic nature of both sulfuric acid and sulfate salts, sulfate aerosols are prone to grow by accumulation of water, so that their effective optical cross section is enhanced far beyond the actual sulfate mass. Since sulfate aerosols are very efficient at scattering light, their impact on Earth's radiation balance is predominantly negative forcing due to backscatter of incoming solar radiation; this effect may be enhanced if their hygroscopicity contributes to increased daytime clouds or fog, or may be somewhat offset if they increase the presence of nighttime clouds or fog. The importance of pollutant sulfate in climate was only fully appreciated in the last decade; inclusion of sulfate cooling helped to significantly reduce the gap between climate change predicted based on GHG calculations and observed secular temperature records (Charleson, et al., 1992). Future reductions in global pollutant sulfur emissions (necessary to manage impacts on public health and prevent "acid rain") may accelerate climate warming as the artificial cooling effect of sulfate is removed (IPCC, 2001).

The precursor sulfur compounds come from both natural and anthropogenic sources.

8.4.1.1 Natural Sulfate

Globally, most natural sulfate comes from biogenic production (primarily in the oceans), with volcanic emissions contributing modestly (e.g., hot springs and fumaroles) on a continuing basis, and occasionally very intensely (large eruptions). As a result, natural sulfate concentrations are somewhat higher over the oceans and lower over the continents. This tends to focus sulfate effects, suppressing solar input to the oceans (lowering heating and evaporation) while minimally altering radiation balance over continents. Large volcanic eruptions have been observed to cool the globe for months or years, an effect believed to be largely due to sulfate. Natural sulfate levels in the atmosphere have been estimated from observations and calculation of emissions, and their climatic effect estimated as well (Twomey, 1974; Twomey, 1977; Charleson, 1987).

8.4.1.2 Anthropogenic Sulfate

Anthropogenic sulfate is generated through the same pathways, but the precursor gases generally come from sulfur bound in fuels used in combustion processes (predominantly coal and petroleum). The potential effects of anthropogenic sulfate are strongest near industrialized regions where large amounts of fossil fuels are burned, thus the cooling effect is strongest over eastern North America, Europe, eastern Asia, and the oceanic and continental areas downwind of these regions (Charleson, 1992; IPCC, 2001a).

8.4.2 Nitrate

Nitrate aerosols form analogously to sulfate, and have similar optical properties. They are distinct from sulfate, however, in that nitrate salts are unstable and can return to the vapor phase when humidity drops or the surrounding air's concentration of precursor gases drops. The dynamics of nitrate aerosol formation and disappearance limit the scope of nitrate impacts on global climate processes.

Nitrates may play an important role on a local or regional basis, especially if their effect is amplified by contributing to changing fog frequency or persistence. Nitrates may be important in some regions as a damper on total aerosol reductions from sulfur control: sulfuric acid has a greater affinity for ammonia than does nitric acid, thus, in a region rich in both SOx and NOx, reducing sulfur emissions may not reduce total aerosol concentrations as nitrate replaces sulfate under humid conditions.

8.4.3 Carbon

Carbonaceous aerosols primarily come from incomplete combustion of fuels, consisting of pure unburned "elemental" carbon ("soot"), partially oxidized organic compounds, and some associated inorganic material ("ash"). In addition, some organic aerosols are produced by gas-phase oxidation of organic vapors – referred to as "secondary" organic aerosol. Carbonaceous aerosols can exhibit highly varied optical effects depending on particle size and chemistry. Major global sources of carbonaceous aerosols are biomass burning (wild fires, vegetation clearing, agriculture, and wood and charcoal used as domestic fuels), industrial and utility boilers, and motor vehicles. Global data on total carbonaceous aerosol emissions are highly uncertain, due primarily to the difficulty of accounting for biomass burning. Moreover, even when current biomass emissions are known, the task will remain to isolate the role of humans in both the amount of burning we initiate and the changes in global biomass fuel patterns wrought by human alteration of the landscape.

Since carbonaceous aerosol emissions are closely linked with CO_2 emissions, properly calculating the aerosol effects alone may be misleading, since any effort to modify these emissions will undoubtedly be linked with significant changes in CO_2 emissions as well. Overall, the effect of carbonaceous aerosol is thought to be positive forcing, but the size of the effect and its regional distribution are highly uncertain.

8.4.3.1 Elemental Carbon

Elemental carbon (EC) aerosols strongly absorb light at all wavelengths, as well as scattering light in wavelengths near the size of the particles. EC's broad-spectrum light absorption gives it a strong potential for positive radiative forcing since it directly absorbs incoming sunlight, turning it into heat in the air containing the aerosol.

EC is produced in almost all combustion processes. The EC fraction of carbonaceous emissions is small in well-controlled fossil fuel combustion, with the notable exceptions of uncontrolled diesel engines, older jet engines, and open burning of oil-based fuels (e.g., burning contaminated waste fuel).

Biomass EC is highly uncertain, in part due to the lack of data on burning activity, and to the fact that the EC fraction is variable depending on fuel moisture and plant species. However, measurements have shown EC to be only about ten percent of biomass aerosol, suggesting that its effects would be overwhelmed by those of the OC and ash content.

8.4.3.2 Organic Carbon

Organic carbon (OC) aerosols generally exhibit a strong wavelength bias in absorption, weak in visible wavelengths and peaking in the ultraviolet. Since the peak of solar energy input is in the visible wavelengths, scattering of visible light has a greater effect on energy balance than UV absorption, thus OC aerosols' climate effects are believed to be weak negative forcing. OC aerosols are often part of a complex mixture ("smoke"), including OC, ash, and water. Because the inorganic fraction of smoke aerosols are

generally weak absorbers at all wavelengths, and the entire mass is capable of scattering light, "smoke" aerosols are considered to show weak negative radiative forcing.

8.4.4 Mineral Dust

"Mineral dust" is generally derived from soil surfaces, either as a result of natural or anthropogenic causes. Since only particles with relatively long atmospheric lifetimes contribute significantly to global aerosol loading, mineral dust at the global scale is quite different from the dust air pollution regulators commonly encounter close to a source. Near-source mineral dust is composed of a variety of crystalline materials, including sand, fine rock fragments ("silt"), and clay particles. Sand and silt materials such as silica have high specific densities and generally fracture into compact shapes, thus coarse mineral particles (>5μm diameter) settle rapidly and have very short atmospheric lifetimes. Conversely, clays, having sheet crystal structures and much smaller particle dimensions, have very large surface to mass ratios and very small settling velocities. Global "background" mineral aerosol is thus finer (mass median diameter near 2μm) and often chemically distinct from most local-source mineral PM.

The optical properties of global mineral aerosols are not well known, nor are their global distributions. Mineral dust may cause either positive or negative radiative forcing, depending on chemistry (fraction of light absorbing minerals) and size (fines scatter more efficiently) (Tegen & Lacis, 1996; Alpert, et al., 1998). Seasonality of dust emission may also play a role in determining net climate effect by altering the albedo of snow and ice or by positive or negative feedbacks with seasonal temperature cycles.

Mineral dust emissions are moderated by soil condition, plant cover, wind speed, soil wetness, and other factors. Human disturbance of soil can greatly increase dust emissions, both directly (tillage) and indirectly (overgrazing, ground water withdrawal, etc.) (Tegen et al., 1996). The fraction of global dust that is due to current human activity is highly uncertain. As with the biomass problem, determining a "natural" (no human effects) baseline will require unraveling the history of human land use and vegetation change as well as compiling emission inventories.

8.5 Vegetation and Materials Damage

The chemical diversity of particulate matter in the air gives is the potential to have a wide range of interactions with surfaces or water bodies on which it deposits. The most significant of these depositional effects involve the acid ions (primarily sulfuric and nitric) within the aerosol. Acid deposition occurs when aerosols or precursor gases deposit on leaves, soil, water, buildings, or other surfaces. Other components of PM also have deleterious effects, primarily in the form of soiling, and, in the cases of certain localities or particularly sensitive "receptors," damaging effects ranging from crop damage to deterioration of water quality.

8.5.1 Acid Deposition Programs

Nitrogen-containing gases and particles are the greatest source of airborne acidity in California. This is in sharp contrast to the eastern United States (U.S.), where precipitation chemistry is dominated by sulfur-containing acids. Nitrogen-containing acids are responsible for a major portion of acidity in precipitation, fogs and clouds, dry deposited gases, and particles within the state. Although annual precipitation acidity is ten-fold lower in California than in the eastern U.S., summertime concentrations and deposition of nitric acid vapor and particle nitrate are among the highest in the nation. While acute, short-term effects on human health and welfare (*i.e.*, agricultural crops and

man-made materials) were determined to be minor, long-term effects on human health, as well as aquatic and forest ecosystems, remain poorly known.

In 1980, the National Acid Precipitation Assessment Program (NAPAP) was established to investigate the causes and effects of acidic deposition in the U.S. While the cause of acidic precipitation is largely due to the dissolution of sulfur and nitrogen oxides in rain, the impacts of sulfur-derived acids were of principal concern in the eastern U.S., and the effects of nitrogen-derived acids were of primary interest in the western U.S. In consideration of the nitrogen-dominated rain chemistry of California, and the potential for distinct health and welfare effects from the eastern U.S., two five-year programs of monitoring and research were enacted by the California Legislature: the Kapiloff Acid Deposition Program (KADP) and the Atmospheric Acidity Protection Program (AAPP). Concentrations of acidic air pollutants in precipitation, fog, and dry-deposited particles and gases were measured in support of the KADP and AAPP by the Air Resources Board's (ARB) California Acid Deposition Monitoring Program (CADMP). Analyte levels in rain/snow and dry deposition have been reported in data summaries (Takemoto et al., 1996), final reports (Watson et al., 1991; Blanchard and Michaels, 1994), and the open literature (Blanchard and Tonnessen, 1993; Melack and Sickman, 1997). The major findings from the KADP and AAPP have also been documented in final reports, Annual Reports to the Governor and the Legislature (ARB, 1983-1986; 1988; 1991-1994a), a technical assessment (ARB, 1989), and the open literature (*e.g*., Takemoto et al., 1995).

8.5.2 Deposition

8.5.2.1 Acidity

Across the state the deposition of N-derived acidic gases and particles provides most of the atmospheric acidity and N to urban landscapes, and to mid-elevation forests in southern California. Blanchard et al. (1996) used precipitation chemistry data from the CADMP, the National Atmospheric Deposition Program/National Trends Network (NADP/NTN), and an alpine precipitation sampling network in the Sierra Nevada Mountains to estimate regional-scale rates of wet-deposited nitrate, sulfate, ammonium, calcium, and H^+ from 1985 through 1994 (Figure 8.5). Rates of wet sulfate, nitrate, and ammonium deposition were found to be <4, <3, and <4 kg S or N/ha/yr in at all sites, respectively (Blanchard et al., 1996). In comparison, rates of wet sulfate and nitrate deposition in eastern North America exceed 8.3 and 3.4 kg S or N/ha/yr, respectively, and deposition rates of ammonium are <3.1 kg N/ha/yr (Sisterson, 1991). In most years, wet nitrate deposition was estimated to be greater in urban areas of the South Coast Air Basin (SoCAB) and the southern Sierra Nevada, than in other parts of California. Along the northwest coast where wet sulfate deposition is highest, much of the sulfate is derived from sea salt. Uncertainties in the wet deposition estimates are #20 percent in the SoCAB, which has a large number of monitors, but are two to three fold higher in other parts of the state.

Comparisons of estimated NO_x emission and total N deposition rates (wet and dry) show that the deposition of oxidized N in the SoCAB accounts for 16-37 percent of the NO_x emitted in the Basin (Figure 8.6; Blanchard et al., 1996). The total N deposition at Fremont was about 11 percent of the NO_x emission rate in San Francisco Bay Area. Total N deposition rates at Bakersfield and Sacramento are about 76 and 32 percent of the NO_x emission rates in Kern and Sacramento County, respectively. Transport of NO_x from upwind areas could account in part for the relatively large deposition-to-emissions ratio at Bakersfield (Tracer Technologies, 1992).

8.5.2.2 Particulate Matter Concentrations

The CADMP dry deposition monitoring program was established in 1988 to determine spatial and temporal patterns of acidic pollutant concentrations in the state. Daytime and nighttime dry particle and gas concentrations were measured once every six days (Watson et al., 1991). Initially, the network consisted of ten sites located in Azusa, Bakersfield, Fremont, Gasquet, Long Beach, Los Angeles, Sacramento, Santa Barbara, Sequoia National Park, and Yosemite National Park. Over the years, data analyses indicated that acidic pollutants were a moderate-to-minor problem in California, and the number of monitoring sites was reduced, as well as the frequency and range of pollutants sampled. In September 1995, the CADMP dry deposition network was reduced to five sites (Azusa, Bakersfield, Long Beach, Los Angeles, and Sacramento) in urban areas. Also, instead of collecting daytime and nighttime samples of PM10 and PM2.5, only one 24-hour-average sample of PM2.5 was collected.

From 1989-1994, annual-average PM10 and PM2.5 concentrations declined at all ten sites. Representative data from five sites are shown in Figures 8.5 and 8.6. Most areas with high PM10 levels also have high PM2.5 concentrations. At rural sites (Gasquet, Yosemite, and Sequoia National Parks), annual average concentrations of PM2.5 were

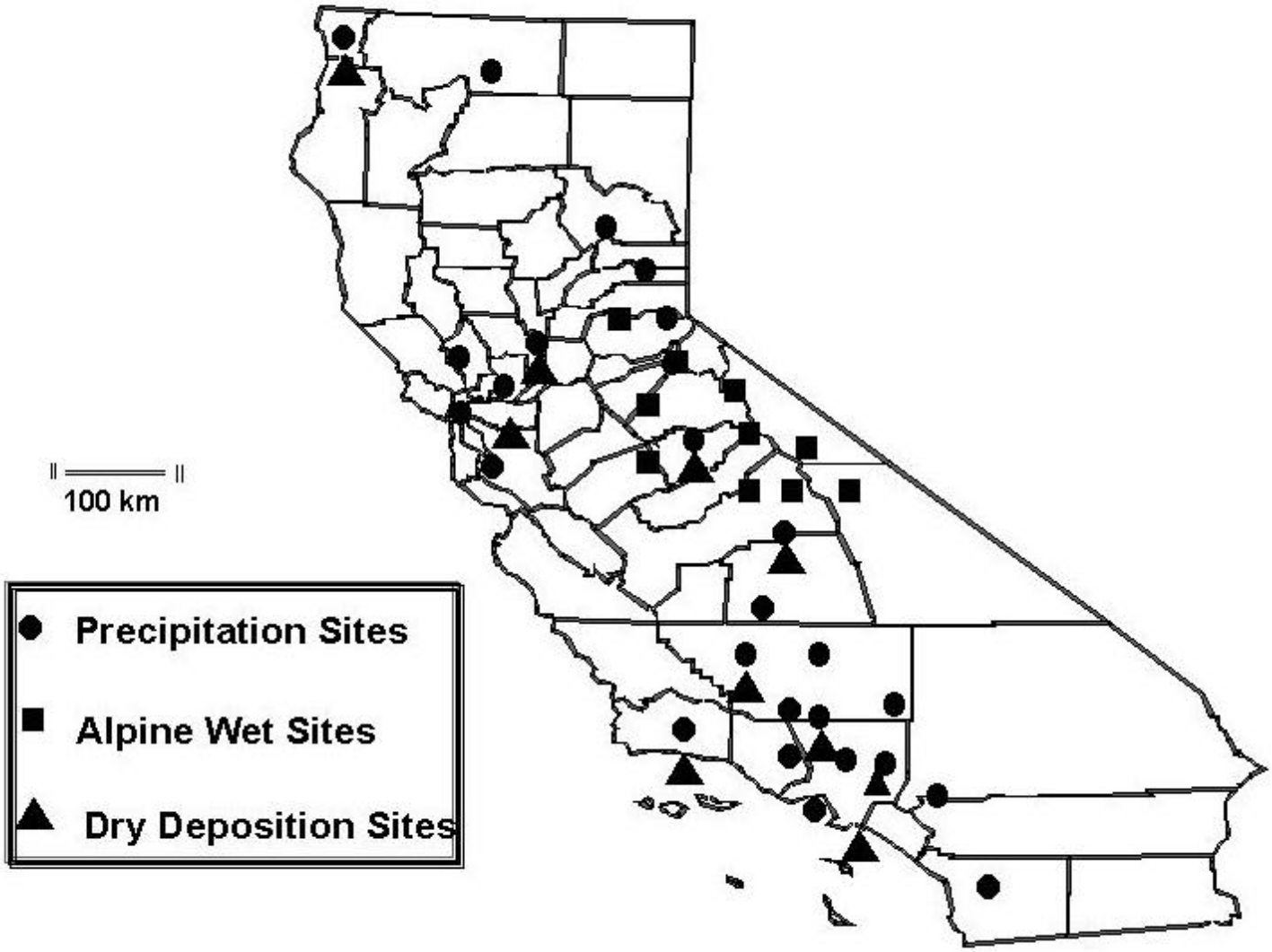

Figure 8.5 Location of CADMP, NADP/NTN, and Sierra Nevada Alpine Wet Deposition Monitoring Sites (Air Resources Board, 1983).

4-6 $\mu g/m^3$. Near to Redwood National Park, Gasquet is far removed from most anthropogenic emissions sources, and provides an estimate of background ambient PM

concentrations in California. On the western slope of the Sierra Nevada, Sequoia and Yosemite National Parks receive pollutants transported from the San Joaquin Valley by upslope flows. Compared to these rural sites, annual-average concentrations of PM2.5 are two to five times greater at urban locations.

8.5.2.3 Acid Fog

Acidic fog has been associated with harmful air pollution episodes (e.g., London, the Meuse Valley in Belgium, and Donora, Pennsylvania), and reported to adversely affect materials, crops, and forests. From 1982 through 1989, ARB sponsored fog water sampling programs at seven sites in California. Fog water collected in the western portion of the SoCAB was found to be highly acidic, with pH values ranging from 1.7 to 4 (e.g., Jacob et al., 1985). Fog water collected at non-urban, coastal sites was less acidic (i.e., pH ranged from 3 to 7) due, in part, to the low alkalinity of marine atmospheres. In the eastern part of the SoCAB and the southern San Joaquin Valley, fogs were generally not as acidic due to high levels of acid-neutralizing ammonia.

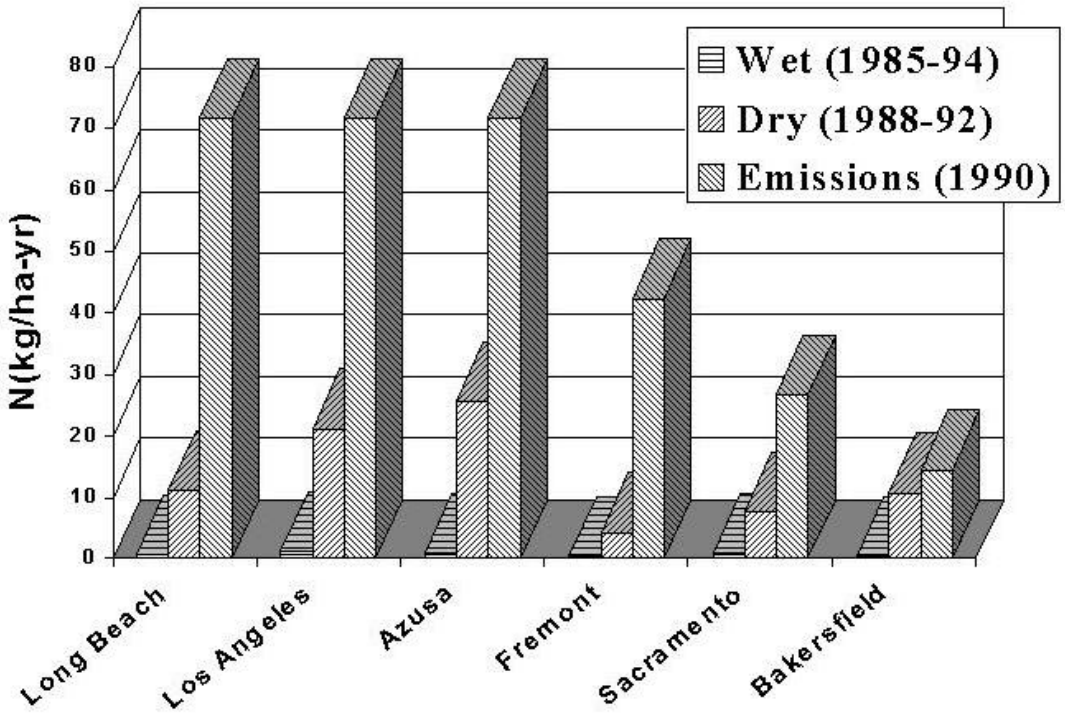

Figure 8.6 Rates of Oxidized N Emissions, and Wet and Dry N Deposition at Urban CADMP Sites (cf. Blanchard et al., 1996).

As in rain, the main contributors to fog acidity are nitric and sulfuric acid. Across the state, the nitrate-to-sulfate ratios in fog are typically about 3:1, but local emissions influence measured concentration ratios. For example, the 3:1 ratio typifies areas where motor vehicle emissions of NO_x dominate (e.g., Los Angeles), but may be close to 1:1 at sites in the southern San Joaquin Valley where sulfur emissions from oil production are significant. Concentrations of ammonium, nitrate, and sulfate ions are commonly 100-times higher in fog than in rain. High concentrations of chemical components in fog correlated well with the occurrence of photochemical smog events, as well as the physical processes of condensation and evaporation.

8.5.3 Effects

In this section the major findings from six research programs sponsored under the KADP and AAPP are summarized. These studies examined the atmospheric processes associated with acid deposition and its effects on human health, aquatic ecosystems, forest ecosystems, agricultural crops, and man-made materials. Statewide networks to monitor pollutant concentrations in wet and dry deposition were established to measure conditions in both urban and rural areas.

8.5.3.1 Aquatic Environments

Changes in surface water chemistry and precipitation chemistry may cause ecosystem-level alterations in the high elevation watersheds of the Sierra Nevada. Chronic acidification of high elevation surface waters in the Sierra Nevada has not been found, but episodic depressions in acid neutralizing capacity do occur. While no large-scale or widespread adverse ecological impacts have been detected, many high elevation aquatic ecosystems are nitrogen-limited and potentially at risk from current levels of atmospheric nitrogen deposition.

Currently, surface waters in the Sierra Nevada are not acidic enough to threaten the juvenile or adult stages of Sierra Nevada amphibians or fish. Of the five species of trout found at high elevation in the Sierra Nevada, three species spawn in the spring (rainbow, golden, and cutthroat), and two spawn in the fall (brown and brook). As a result they are differentially at risk from episodic acidification (Jenkins et al., 1994). In spring, the fertilized eggs of spring-spawning trout are at risk from snowmelt water, which is considerably more acidic than pre-melt surface water.

Episodic acidification of streams due to snowmelt or summer rains may decrease populations of some species of stream invertebrates. Vulnerable species identified in work done at Emerald Lake include the nymphs of mayflies and chironomid fly larvae (Hopkins et al., 1989; Kratz et al., 1994). When pH is lowered to 5.0 or below, for as little as eight hours, drift rates of vulnerable species increase, and much of the increased drift is due to mortality (i.e., drifting insects are killed by low pH).

Using the 1985 USEPA Western Lakes Survey, it was estimated that none of the 114 lakes sampled in the Sierra Nevada had been episodically acidified (ANC < 0) (Leydecker et al., 1999). These workers predicted that approximately six and ten percent of Sierra lakes would become episodically acidified if nitrate and sulfate deposition increases by 50 and 150 percent, respectively. No lakes would be chronically acidified in response to the above increases in nitrate and sulfate deposition.

In Lake Tahoe, studies (Jassby, et al, 1994) indicate that phytoplankton growth is not co-limited by the availability of nitrogen and phosphorus; rather, growth is limited by phosphorus alone, due to the deposition of atmospheric nitrogen. Nutrient input to Lake

Tahoe, including airborne nitrogen and phosphorus, is not only a concern for ecosystem effects, but is believed to be a major factor in loss of clarity in the lake.

8.5.3.2 Forests

Nitrogen saturation has occurred in forested watersheds in the San Bernardino Mountains, and nitrate contamination of groundwater is of near-term concern. In future years, atmospheric nitrogen deposition could lead to forest soil nitrogen saturation in other areas such as the San Gabriel Mountains and southern Sierra Nevada. Ozone is the primary air pollution stressor of forests, and there is the potential for interactive effects with atmospheric nitrogen.

8.5.3.3 Crops

The acute effects of acidic fog on crops were of concern in the 1980s following reports of adverse S-derived fog and aerosol effects on human health (Graham, 1991). Two studies were funded to evaluate effects on winter and summer crops (Olszyk et al., 1987), and two species of conifer seedlings (Bytnerowicz et al., 1989). As the most extreme fog exposure, a pH 1.7 fog treatment was applied to simulate the pH 1.69 fog measured in Corona Del Mar by Hoffman and co-workers at the California Institute of Technology (Jacob et al., 1985). The responses of five crops were examined, and four crops exhibited yield reductions following 11 weeks of exposure to pH 1.7 fog (Olszyk et al., 1987). The damage to leaves caused by pH 1.7 fog decreased the amount crop leaf area capable of performing photosynthesis. The observed reductions in crop yield were largely explained by decreases in whole plant photosynthesis. Similar findings were reported for white fir and ponderosa pine seedlings exposed to pH 2.0 fog for six weeks (Bytnerowicz et al., 1989).

8.5.3.4 Soil Chemistry

Concern over the effects of acidic deposition on agricultural soils emerged as a result of findings that suggested that excess inputs of N and S could lead to trace element nutrient deficiencies (*e.g.*, calcium). In a report by Mutters (1995), the nutrient requirements of selected crops were compared against annual inputs from fertilizer and the atmosphere to determine if imbalances could develop. Of the three elements examined (N, S, and calcium), there was a limited possibility that atmospheric N deposition could contribute to a build-up of nutrients that could adversely affect crop productivity. Given the lack of direct acidic deposition impacts on crop growth or yield, no additional research is needed. In terms of ARB's air quality goals, current farm practices appear to provide adequate protection from the harmful effects of acidic deposition.

8.5.3.5 Man-made Materials

Studies conducted in both the KADP and AAPP did not identify any significant damage to materials due to atmospheric acidity. While laboratory analyses indicate that NO_2 and nitric acid may damage painted surfaces, aluminum, and nylon fabric (Mansfeld et al., 1988), field studies in southern California found corrosion rates to be similar to rates in sites with clean air (Mansfeld and Henry, 1993).

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9 Controls and Reduction Strategies for Particulate Matter

Particulate matter (PM) can be either directly emitted into the air in forms such as dust and soot, or it can be formed in the atmosphere (like ozone) from the reaction of gaseous precursors such as nitrogen oxides (NOx), volatile organic compounds (VOCs), sulfur oxides (SOx), and ammonia. Directly emitted particles are called "primary particles," while those formed in the atmosphere are referred to as "secondary particles." Both primary and secondary particles contribute to PM episodes in California, with some episodes dominated by primary particles and others dominated by secondary particles. Therefore, control programs to reduce particulate matter levels must address both primary and secondary particles. Generally, ARB or U.S. EPA control emissions from mobile sources such as cars, heavy trucks and off-road equipment, and local air districts control emissions from stationary or area sources.

The federal Clean Air Act requires federal PM10 nonattainment areas that have been classified "moderate" to implement Reasonably Available Control Measures (RACM), while "serious" nonattainment areas must implement Best Available Control Measures (BACM). Under the California Clean Air Act, air districts must attain the more stringent state PM standards as expeditiously as practicable.

For RACM and BACM, U.S. EPA guidance for controlling small or dispersed source categories such as road dust, woodstoves, and open burning defines the "reasonable" or "best" control measures. Primary PM control measures include application of dust suppressants such as chemicals or water, use of coverings or enclosures to prevent dust from escaping into the ambient air, planting open areas to stabilize the soil, paving of dirt roads, and reduction of vehicle speeds on dirt roads. BACM is more stringent than RACM and results in greater control efficiencies. For example, BACM for unpaved roads may consist of more miles of road to be paved than RACM. Because many of the same pollutants that form ozone also form secondary particulate matter, ozone reductions programs also reduce emissions of PM precursors.

The following sections describe some of the PM control programs currently in place in California. In ARB's Clean Air Plan, which is scheduled to be considered by the Board in the first half of 2002, we will describe our future plans to control both primary sources of particulate matter and particulate matter precursors to further reduce ambient levels of particulate pollution.

9.1 Primary Particle Control

9.1.1 Fugitive Dust Rules

Fugitive dust is generated from activities that disturb the earth's surface area. Some of these activities include: travel on unpaved roads, windblown dust from open storage piles of materials such as soil or animal feed, carryout of dirt from unpaved areas that adheres to vehicles or equipment and falls onto paved roads (also known as "trackout"), or earth-moving activities such as grading or landfill operations.

Several districts have adopted fugitive dust rules to reduce particulate matter using RACM and BACM (as described above). ARB also continues to work with the districts in conducting research for source identification and development of additional control measures for fugitive dust.

9.1.2 Burn Rules

Several different burn rules have been implemented to help reduce the particulate emissions that are a result of all burning activities.

9.1.2.1 Agricultural and Prescribed Burning

Agricultural burning refers to the intentional use of fire for removal of vegetative waste, disease and pest prevention, forest operations, and range improvement in areas such as agricultural fields, orchards, and wildlands. Agricultural burning also includes "prescribed burning," which are fires intentionally ignited to meet specific land management objectives.

California agricultural burning guidelines were established in 1971 to reduce the harmful health effects caused by particulate matter and gases in smoke from unrestrained open burning on public and private lands. In March of 2000, ARB amended the regulations to allow for necessary open burning, while still protecting public health and air quality. The regulatory changes improve interagency coordination and ensure use of real-time meteorological data to avoid smoke episodes. ARB staff is currently evaluating the need for additional modifications to the smoke management regulations, which would ban all agricultural and prescribed burning on days forecast to violate an ambient air quality standard.

9.1.2.2 Indoor Residential Wood Combustion

In 1989, ARB developed a Suggested Control Measure to assist districts in emission reductions from residential wood combustion. Since then, a few districts have developed residential wood burning rules. Control strategies include mandatory phase-out or change-out of fireplaces, voluntary burning reduction on days with poor air quality, and public awareness and education from device suppliers.

ARB is investigating the need for and feasibility of additional control measures, including building incentive programs, and strengthening existing public awareness and education programs.

9.1.2.3 Outdoor Residential Waste Burning

In parts of California, some types of residential solid waste, such as household garbage, paper products, wood products, cloth, and plastics are allowed to be burned outdoors. In addition to the typical gaseous and particulate matter emissions from burning activities, burning of residential solid waste also results in the release of toxic air contaminants such as dioxin, benzene, and 1,3-butadiene. ARB staff is developing an Airborne Toxics Control Measure (ATCM) which would then be adopted and implemented by local air districts to address this issue.

Staff is considering a variety of options, including banning residential burning of all waste except vegetation (with limited exceptions in areas lacking waste disposal options) and banning the use of burn barrels. This would be coupled with encouraging improvements to waste collection and disposal infrastructure, and facilitating compliance through increased public education and outreach. Reduced burning of residential waste will reduce direct PM as well as toxics.

9.1.3 Diesel Reduction Plan

Both the nitrogen oxides and direct particulate matter emitted by diesel engines contribute to ambient PM levels. In addition, ARB identified diesel particulate matter as a toxic air contaminant in August of 1998.

In September 2000, the ARB approved a comprehensive Diesel Risk Reduction Plan to reduce diesel PM emissions from both new and existing diesel-fueled engines and vehicles. The goal of the Plan is to reduce diesel PM emissions and the associated health risk by 75 percent in 2010 and by 85 percent or more by 2020.

The Plan identifies 14 measures that will be developed over the next several years to: establish more stringent emission standards for new diesel-fueled engines and vehicles; establish particulate trap retrofit requirements for existing engines and vehicles where traps are determined to be technically feasible and cost-effective; require the sulfur content of diesel fuel to be reduced to enable the use of advanced diesel PM emission controls; and evaluate alternatives to diesel-fueled engines and vehicles.

The ARB recently verified the first diesel particulate filters for use in many on-road applications. Under ARB regulations, transit buses are already required to begin using diesel particulate filters in 2003. The Board will consider the first new measure under the Diesel Risk Reduction Plan -- requiring diesel particulate filters on new and in-use trash trucks – in mid-2002.

9.2 Secondary Particle Control

To reduce levels of secondary particulate matter, control programs focus on reducing the precursors that form secondary particles: NOx, VOCs, SOx, and ammonia. Because ozone and particulate matter pollution are caused by many of the same sources and precursors (NOx and VOCs are also ozone precursors), many of California's ozone control strategies provide dual benefits for public health by reducing particulate matter as well. In addition to the ARB programs described below, local air districts have programs for stationary and area sources that also reduce emissions of particulate matter precursors.

9.2.1 Clean Fuels

ARB's clean fuels program provides the dual benefit of immediate emission reductions from vehicles currently on the road while also enabling new vehicles to meet increasingly more stringent standards. ARB's cleaner burning gasoline and diesel fuel regulations have significantly reduced emissions of VOCs, NOx, SOx, PM, CO, and air toxics.

9.2.2 Mobile Source Program

ARB's mobile source control program, in tandem with fuel standards, is the cornerstone of our strategy for achieving clean air. Since the 1960s, ARB has adopted emission standards for virtually every category of on- and off-road motor vehicle, including cars and trucks, heavy-duty diesel trucks, lawnmowers and other garden equipment, and the engines used in off-road equipment such as tractors, backhoes, and portable generators. ARB's clean vehicle strategy relies on three basic elements: requiring new engines to be as low-emitting as possible, ensuring that those engines maintain their low emissions while they are on the road, and encouraging the retirement of older, more polluting engines. The program has resulted in dramatic reductions in vehicle emissions, despite continued increases in population and vehicle travel.

9.2.3 Consumer Products Program

Since 1989, ARB has adopted emission limits for over 80 categories of consumer products and aerosol paints, resulting in a nearly 20 percent reduction in VOC emissions from these products.

9.2.4 Vapor Recovery Program

Vapor recovery systems are used to capture VOCs during the transfer of gasoline between storage tanks and tanker trucks as well as the refueling of vehicles at gasoline pumps. For over 20 years, ARB has adopted the certification and test procedures to ensure vapor recovery systems meet minimum standards.

10 Quantifying the Adverse Health Effects of Particulate Matter

This chapter reviews the process used to estimate the health effects of particulate matter (PM) in California. It describes the methods used in estimating the adverse health effects, discusses issues that arise with the choice and use of concentration-response (C-R) functions, and presents detailed estimates of annual incidences of the adverse health effects.

10.1 Health Effects Estimation Approach

Estimating the incidence of adverse health effects of PM involves four elements:

- Estimates of the changes in PM exposure levels.
- Estimates of the number of people exposed to PM at a given location.
- C-R functions that link changes in PM concentration with changes in the incidence of adverse health effects.
- Applicability of the C-R functions that are drawn from studies conducted in other parts of the country to California.

Each of these elements is discussed below.

10.1.1 Exposure Estimation and Assumptions

The basic procedure for determining exposures was first adopted by the ARB in 1993 to fulfill the requirements of Section 39607(f) of the Health and Safety Code. Full details are provided in Guidance for Using Air Quality-Related Indicators in Reporting Progress in Attaining the State Ambient Air Quality Standards (September 1993). For this application, the concentrations and populations were associated by census tract and merged to assemble a distribution of exposures to different concentrations of PM.

Concentrations of many air pollutants, including PM, change significantly from one location to another. PM concentrations may be well under the standard in one location but above the standard less than 10 kilometers away. Accordingly, population exposures tend to be more accurate when the population data are highly resolved.

Population counts by census tract are used to determine population exposures to air pollutants. In addition, demographic data, such as age distributions, are available for each census tract. A typical census tract contains several thousand people. Densely populated areas have many census tracts, while sparsely populated regions have few.

We estimated PM10 and PM2.5 concentrations per census tract using air quality data from monitors located at specified distances from the census tract centroid. Air pollutant concentrations from a network of air quality monitors are used to determine appropriate values at census tracts that lie between the monitors.

The concentration for a census tract is the weighted average of the concentrations at all monitors within a maximum allowed distance. For the present analyses of PM10 and PM2.5, the maximum distance was 50 kilometers except for 75 km in the Great Basin Valleys Air Basin. A small number of census tract populations were not included in the analyses because they are more than 50 km from any PM monitor. The population numbers are affected only slightly by different choices for the maximum distance.

The weight assigned to each monitor is the inverse square of its distance from the census tract. In this way, close monitors are more influential than distant ones. Although "boundaries," such as mountain ranges, were not used in the model, local monitors on each side of such boundaries dominate the calculated concentrations for census tracts in their respective regions.

In each air basin, we assumed that the population in a specific concentration bin is exposed to the mid-point concentration in that bin. We then estimated the population-weighted PM2.5 and PM10 annual arithmetic mean concentration in each air basin.

10.1.2 Data Used

Monitoring data for 1998 through 2000 were used from all monitors in the State meeting quality assurance criteria for valid data. Projected census tract data based on 1990 census data were used as the 2000 data were not yet available in the census tract format. The census data contains the shape, size and centroid of each census tract.

10.1.3 Exposure Model Results

Table 10.1 summarizes the results of the statewide assessment.

Table 10.1. Population-Weighted Average Particular Matter Annual Arithmetic Mean Concentration

Air Basin	PM2.5 (ìg/m3)	PM10 (ìg/m3)
Great Basin Valleys	8.50	16.71
Lake County	2.50	10.83
Lake Tahoe	7.50	20.83
Mountain Counties	16.60	22.96
Mojave Desert	10.00	21.60
North Coast	7.50	17.54
North Central Coast	7.50	24.25
Northeast Plateau	NA	12.97
South Coast	22.20	40.67
South Central Coast	11.80	23.04
San Diego	15.60	28.80
San Francisco Bay Area	15.80	21.67
San Joaquin Valley	22.30	39.48
Salton Sea	13.10	70.17
Sacramento Valley	12.30	24.49
Statewide Averages	18.5	33.11

10.1.4 Exposed Population by Location

Health effects are related to the level of PM that individuals are exposed to. Because the levels of PM exposure vary from air basin to air basin, individuals in different air basins do not experience the same health effects. Estimating health effects by county is complicated somewhat because concentrations were estimated by air basin rather than by county in this analysis. The boundaries for air basins and counties are not always the same due to geographic characteristics. Therefore, county populations were divided to fit air basin boundaries.

We estimated the basin county population, i.e., the county population within an air basin, based on the county population percentage relative to the air basin population derived from California Department of Finance air basin population data and the 2000 census county population.

10.1.5 Concentration-Response Functions

Concentration-response (C-R) functions are equations that relate the change in the number of adverse health effect incidences in a population to a change in pollutant concentration experienced by that population. This section discusses issues that affect health effect estimates and outlines epidemiological studies used for the basis of the C-R functions. Many C-R functions were used in the U.S. EPA Final Heavy-Duty Engine/Diesel Fuel Rule: Air Quality Estimation, Selected Health and Welfare Benefits Methods, and Benefit Analysis Results.

10.1.5.1 Basic C-R Function

Different epidemiological studies have been used to estimate the relationship between PM and a particular health endpoint at different locations. The C-R functions estimated by these studies may have different functional forms, PM concentrations, health endpoints, and relate to different populations.

Some studies have assumed that the relationship between a health endpoint and PM is best described by a linear form, i.e., the relationship between a health endpoint (Y) and PM is estimated by a linear regression in which Y is the dependent variable and PM is one of several independent variables. Other studies have assumed that the relationship is best described by a log-linear form, i.e., the relationship between the natural logarithm of Y and PM is estimated by a linear regression. Most common functions used in this analysis are in log-linear form with a few exceptions using logistic regressions.

A log linear C-R function is:

$$\Delta y = y_0 \ (e^{-\hat{a}\Delta PM} - 1) \cdot pop$$

where:

Äy = changes in the incidence of a health endpoint corresponding to a particular change in PM in a population

y_0 = baseline incidence rate per person

â = coefficient

ÄPM = change in PM concentration

pop = population of a particular group that a study considered.

The parameters in the functions differ depending on the study. Some studies of the relationship between ambient PM concentrations and mortality have excluded accidental

deaths from their mortality counts; others have included all deaths. Some studies considered only members of a particular subgroup of the population, e.g., individuals 65 and older while other studies considered the entire population in the study location. When using a C-R function from an epidemiological study to estimate changes in the incidence of a health endpoint corresponding to a particular change in PM in a location, it is important to use the appropriate value of parameters for the C-R function. That is, the measure of PM, the type of population, and the characterization of the health endpoint should be the same as or as close as possible to those used in the study that estimated the C-R function.

10.1.5.2 Baseline Incidences

The health effect baseline incidences are the baseline incidence rate in a specific location multiplied by the relevant population. County mortality rates are used in the estimation of air pollution-related mortality. Hospital admissions are calculated at the state level for a given population age group based on "Patient Discharge Data 1998-1999", California Office of Statewide Health Planning and Development, December 2000. All counties are assumed to have the same incidence rate for a given population age group. For some endpoints, such as respiratory symptoms, respiratory illnesses, and restricted activity days, baseline incidence rates are not available even at the national level. The only sources of estimates of baseline incidence rates in such cases are the studies reporting the C-R functions for those health endpoints.

10.1.5.3 Thresholds

Different assumptions about whether to apply thresholds, and at what levels, can have a major effect on health effects estimates. A very important issue in estimating PM health effects is whether it is valid to apply the C-R functions throughout the range of predicted changes in ambient concentrations, even changes occurring at levels approaching the natural background concentration (without any human activity).

There is some evidence that, at least for particulate matter, not only is there no threshold, but the PM effect coefficient may actually be larger at lower levels of PM and smaller at higher levels (Rossi et al., 1999). However, we used the background concentration of PM as a threshold for estimating the health effects presented in this analysis. As a result, adverse health effects may be underestimated.

The Point Reyes National Seashore in Northern California is located away from populated areas and other significant sources of particulate and particulate precursor emissions. Thus the PM concentration at this site may represent an estimate of PM concentrations in the absence of anthropogenic emissions. Data obtained from the IMPROVE program for Point Reyes from March 1996 through February 1999 indicate that annual average concentrations were 4.55 ìg/m^3 for PM2.5 and 10.97 ìg/m^3 for PM10. In this analysis, we applied thresholds of 5 ìg/m^3 for PM2.5 and 10 ìg/m^3 for PM10 in all the epidemiological functions except for the long-term mortality functions where we used 9 ìg/m^3 for PM2.5 and 18 ìg/m^3 for PM10—the lowest concentration levels observed in the two long-term mortality studies. We assumed that all of these functions were continuous and differentiable down to threshold levels.

10.1.5.4 Mortality

Premature mortality may result from either short-term or long-term exposure to pollution concentrations. Short-term exposure may result in excess mortality on the same day or within a few days of increased exposure. Long-term exposure (over a year or more) may result in mortality in excess of what it would be if PM levels were generally lower. Long-

term exposure may capture a facet of the association between PM and mortality that is not captured by short-term exposure.

Long-term epidemiological studies estimate the association between long-term (chronic) exposure to air pollution and the survival of members of a large study population over an extended period of time. Such studies examine the health endpoint(s) in relation to the general long-term level of the pollutant of concern, for example, relating annual mortality to some measure of annual pollutant level. In contrast, short-term studies relate daily levels of the pollutant to daily mortality. By their basic design, daily studies can detect acute effects but not the effects of long-term exposures. A chronic exposure study design is best able to identify the long-term exposure effects, and may detect some of the short-term exposure effects as well. Because a long-term exposure study may detect some of the same short-term exposure effects detected by short-term studies, a sum of estimated effects from both study types would likely result in some degree of double counting of the effects.

The following four studies were used to estimate PM related mortality.

Long-term Mortality (Krewski et al., 2000) Based on ACS Cohort

This study is a re-analysis of the Pope et al. (1995) study of PM2.5 associated mortality, using American Cancer Society (ACS) data. It essentially confirms the original findings. An advantage of Krewski et al. over Pope et al. is that the reanalysis uses the annual mean PM2.5 concentration rather than the annual median. Because the mean is affected more by high PM values than the median, if high PM days are important in causing premature mortality, the annual mean may be preferable to the median as a measure of long-term exposure. We used this study to derive primary estimates of premature mortality.

The C-R function to estimate the change in long-term mortality is:

$$\text{Ä Mortality} = - y_0 \ (e^{-\hat{a}\text{ÄPM}} - 1) \cdot \text{pop}$$

where:

y_0 = county-level all-cause annual death rate per person ages 30 and older

â = PM2.5 coefficient = 0.0046257, PM10 coefficient = 0.00231285

ÄPM = change in annual mean PM concentration

pop = population of ages 30 and older

$ó_{â}$ = standard error of â PM2.5 = 0.0012046, PM10 = 0.0006023

Incidence Rate. To estimate county-specific baseline mortality incidence among individuals ages 30 and over, we used data from 1999 annual all cause deaths by age by county (Center for Health Statistics, California Department of Health, 1999).

Coefficient Estimate (â). The coefficient (â) for PM2.5 is estimated from the relative risk (1.12) associated with a mean change of 24.5 µg/m^3 (Krewski et al., 2000, Part II - Table 31).

Recent findings reported by Pope et al. of a new analysis of the American Cancer Society data show no association with long-term mortality for coarse (PM2.5 – PM10) (abstract #205, ISEE meetings, 2001). Based on the assumptions that: (1) only PM2.5 (fine PM) is associated with long-term mortality; (2) the reduction in PM10 will maintain the current proportion of PM2.5 in California-suggesting a mix of control strategies; (3)

and state average fine and coarse PM fraction is about 50-50, the coefficient for PM10 was derived by multiplying the PM2.5 coefficient by 0.5. Using this adjusted PM10 coefficient, we only calculated long-term mortality effects for the PM2.5 fraction of PM10. The standard error for PM10 was also adjusted accordingly.

Standard Error ($Ó_{â}$). The standard error for PM2.5 was calculated as the average of the standard errors implied by the reported lower and upper bounds of the relative risk (Krewski et al., 2000, Part II – Table 31).

Long-term Mortality (Krewski et al., 2000) Based on Six-City Cohort by Dockery

Krewski et al., (2000) also reanalyzed the data from another prospective cohort study (the Harvard "Six Cities Study") authored by Dockery et al., (1993). The Dockery et al., study used a smaller sample of individuals from fewer cities than the study by Pope et al., (1995); however, it features improved exposure estimates, a slightly broader study population (adults aged 25 and older), and a follow-up period nearly twice as long as that of Pope et al., We used this study for alternative estimates of long-term mortality effects.

The C-R function is:

$$Ä \text{ Mortality} = - y_0 \ (e^{-âÄPM} - 1) \cdot pop$$

where:

y_0 = county-level all-cause annual death rate per person ages 25 and older

â= PM2.5 coefficient = 0.013272, PM10 coefficient = 0.006636

ÄPM = change in annual mean PM concentration

pop = population of ages 25 and older

$ó_{â}$= standard error of â PM2.5= 0.00407, standard error of â PM10= 0.00204

Incidence Rate. To estimate county-specific baseline mortality incidence among individuals ages 25 and over, we used the data from 1999 annual all cause deaths by age by county (Center for Health Statistics, California Department of Health, 1999).

Coefficient Estimate (â). The coefficient (â) for PM2.5 is estimated from the relative risk (1.28) associated with a mean change of 18.6 (Krewski et al., 2000, Part I - Table 19c). The coefficient for PM10 was adjusted by multiplying the PM2.5 coefficient by 0.5 so that we only calculate a long-term mortality benefit for the PM2.5 fraction of PM10. The standard error for PM10 was also adjusted accordingly.

Standard Error ($ó_{â}$). The standard error for PM2.5 was calculated as the average of the standard errors implied by the reported lower and upper bounds of the relative risk (Dockery et al., 1993, Table 5)

Short-Term Mortality (Schwartz et al., 1996)

Schwartz et al., (1996) pooled the results from six cities in the U.S. and found a significant relationship between daily PM2.5 concentration and non-accidental mortality. Abt Associates, Inc. (1996b, p. 52) used the six PM2.5 relative risks reported by Schwartz et al., in a three-step procedure to estimate a pooled PM2.5 coefficient and its standard error. The first step estimates a random-effects pooled estimate of â; the second step uses an "empirical Bayes" procedure to re-estimate the â for each study as a weighted average of the â reported for that location and the random effects pooled

estimate; and the third step estimates the underlying distribution of â, and uses a Monte Carlo procedure to estimate the standard error (Abt Associates, Inc., 1996a, p. 65).

The C-R function to estimate the change in mortality associated with daily changes in PM2.5 is:

$$\Ä \text{ Mortality} = - y_0 \ (e^{-\hat{a}\ÄPM} - 1) \cdot \text{pop}$$

where:

y_0 = county-level daily incidence for non-accidental deaths per person of any age

â = PM2.5 coefficient (Abt Associates Inc., 1996a, Exhibit 7.2) = 0.001433

ÄPM = change in daily average PM2.5 concentration

pop = population of all ages

$ó_{â}$ = standard error of â (Abt Associates Inc., 1996a, Exhibit 7.2) = 0.000129

Short-Term Mortality (Pooled California PM10 studies, Chestnut, et al., 2001)

A number of daily time-series studies have examined the PM-premature mortality relationship in California populations. Some of the study details and the PM relative risk results from these studies are presented in Table 10.2. Chestnut and Mills pooled PM10 results from each of the counties represented in this table in a random effects model. For counties with more than one set of PM10 results, those estimates were pooled first and the results from a fixed effects assumption were incorporated with the results for the remaining locations. Only PM10 results were used so no results from Kinney and Ozkaynak (1991) or from Ostro (1995) are included in the pooled estimate. The result of the pooled PM10 studies is shown in the last row of the table and it applies to all ages and non-accidental deaths.

Table 10.2. Daily time series study results of impact of PM on daily mortality in California.

Study	Study location (years)	PM measure used in study	Pollutant covariates included	Estimated Beta (std. Err)	Relative risk for a 10 ìg/m^3 (95% CI)
Fairley, 1999	Santa Clara (1989-1996)	PM2.5	Ozone, CO, NO_2	0.004365 (0.001694)	1.045 (1.010, 1.080)
		PM10	None	0.001539 (0.000598)	1.016 (1.004, 1.027)
Kinney and Ozkaynak, 1991	Los Angeles (1970-1979)	KM[b]	Oxides	N/A (linear regression used)	1.008 (1.005, 1.012)
Kinney et al., 1995	Los Angeles (1985-1990)	PM10	Ozone	0.000488 (0.000284)	1.005 (0.999, 1.010)
Ostro, 1995	San Bernardino and Riverside Counties (1980-1986)	PM2.5 (est)	None	0.000000 (0.000311)	1.000 (0.994, 1.006) (full year)

Study	**Study location (years)**	**PM measure used in study**	**Pollutant covariates included**	**Estimated Beta (std. Err)**	**Relative risk for a 10 ìg/m^3 (95% CI)**
Ostro et al., 1999	Coachella Valley (1989-1992)	PM10	None	0.001128 (0.000747)	1.011 (0.997, 1.026)
Samet et al., 2000b	Los Angeles County (1987-1994)	PM10	Ozone	0.000419 (0.000188)	1.004 (1.001, 1.008)
	San Diego County (1987-1994)	PM10	Ozone	0.001124 (0.000467)	1.011 (1.002, 1.021)
	Orange County (1987-1994)	PM10	Ozone	0.001025 (0.000523)	1.010 (1.000, 1.021)
	Santa Clara County (1987-1994)	PM10	Ozone	0.000369 (0.000350)	1.004 (0.997, 1.011)
	San Bernardino County (1987-1994)	PM10	Ozone	0.000310 (0.000687)	1.003 (0.990, 1.017)
	Alameda County (1987-1994)	PM10	Ozone	0.002000 (0.000572)	1.020 (1.009, 1.032)
Random Effects Pooling, Chestnut et al., 2001	**All counties represented in table**	**PM10**	**N/A**	**0.000838 (0.000203)**	**1.008 (1.004, 1.012)**

a. Mortality in these studies is non-accidental mortality, which excludes deaths attributed to homicide, suicide, legal intervention, or other accidental causes.

b. KM is a measure of visual opacity in the air, which is related to particulate matter. The mean value for KM in this study was 25.

10.1.5.5 Chronic Bronchitis (Abbey et al., 1995 and 1993, California)

Abbey et al. (1995) examined the relationship between estimated PM2.5 (annual mean from 1966 to 1977), PM10 (annual mean from 1973 to 1977), and total suspended particulate (TSP, annual mean from 1973 to 1977) and the same chronic respiratory symptoms in a sample population of 1,868 Californians. The initial survey was conducted in 1977 and the final survey in 1987. To ensure a better estimate of exposure, the study participants had to have been living in the same area for an extended period of time. In single-pollutant models, there was a statistically significant PM2.5 relationship with development of chronic bronchitis, but not for airway obstructive disease (AOD) or asthma; PM10 was significantly associated with chronic bronchitis and AOD; and TSP was significantly associated with all cases of all three chronic symptoms.

The C-R function to estimate the change in chronic bronchitis is:

$$\Ä \text{ Chronic Bronchitis} = - y_0 \ (e^{-âÄPM} - 1) \cdot pop$$

where:

y_0 = annual bronchitis incidence rate per person = 0.00378 (Abbey et al., 1993, Table 3)

â = estimated PM2.5 coefficient = 0.0132, PM10 coefficient = 0.00932

ÄPM = change in annual average PM concentration

Pop = population of ages 27 and older without chronic bronchitis = 0.9465*population 27+

$ó_{â}$ = standard error of â = 0.00680 for PM2.5, 0.00475 for PM10

Incidence Rate. The estimation of the incidence rate is detailed in "Final Heavy Duty Engine/Diesel Fuel Rule: Air Quality Estimation, Selected Health and Welfare Benefits Methods, and Benefit Analysis Results, Appendix C", U.S. EPA, December 2000.

Coefficient Estimate (â). The estimated coefficient (â) for PM2.5 is based on the relative risk (= 1.81) associated with 45 µg/m^3 change in PM2.5 (Abbey et al., 1995, Table 2). The estimated coefficient (â) for PM10 is based on the relative risk (= 1.36) associated with 60 µg/m^3 change in TSP (Abbey et al., 1993, Table 5). Assuming that PM10 is 55% of TSP and that particulate greater than 10 micrometers are harmless.

Standard Error ($ó_{â}$). The standard error for the PM2.5 coefficient (â) is calculated from the reported lower and upper bounds of the relative risk (0.98 to 3.25) (Abbey et al., 1995, Table 2).

10.1.5.6 Hospital Admissions

Studies of a possible PM-hospitalization relationship have been conducted for a number of locations in the United States, including California. These studies use a daily time-series design and focus on hospitalizations with a first-listed discharge diagnosis attributed to diseases of the circulatory system (ICD9-CM codes 390-459) or diseases associated with the respiratory system (ICD9-CM codes 460-519). Subcategories within these groups are also often examined, with variation between studies in how the categories are defined. Common subcategories within circulatory are cardiovascular, which includes heart attack, and cerebrovascular, which includes stroke. Common subcategories within respiratory are chronic obstructive pulmonary disease (COPD), asthma, and pneumonia. Various age grouping are also considered, which vary across studies.

Some studies have examined the relationship between air pollution and emergency room (ER) visits. Because most emergency room visits do not result in an admission to the hospital we treated hospital admissions and ER visits separately, taking account of the fraction of ER patients that were admitted to the hospital.

<u>Hospital Admissions for COPD (Samet et al., 2000a, 14 Cities)</u>

Samet et al. (2000a) examined the relationship between air pollution and hospital admissions for individuals age 65 and over in 14 cities across the country. Cities were selected on the basis of available air pollution data for at least four years between 1985 and 1994 during which at least 50% of days had observations between the city-specific start and end of measurements.

The C-R function to estimate the change in hospital admissions for COPD associated with daily changes in PM10 is:

$$Ä \text{ COPD Admissions} = - y_0 \ (e^{-âÄPM} - 1) \cdot pop$$

where:

y_0.= daily hospital admission rate for COPD per person 65 and older = 2.95 E-5

â = PM10 coefficient = 0.00288

ÄPM = change in daily average PM concentration

pop = population age 65 and older

$ó_{â}$ = standard error of â = 0.00139

Incidence Rate. COPD hospital admissions (ICD-9 codes: 490-492, 494-496) are based on "Patient Discharge Data 1998-1999," California Office of Statewide Health Planning and Development, 2000. Population data are from "Race/Ethnic Population with Age and Sex Detail, 1970-2040", California Department of Finance.

Coefficient Estimate (â). The coefficient is estimated from relative risk of 1.029 which is based on a 2.88 percent increase in admissions due to a PM10 change of 10.0 µg/m^3 (Samet et al., 2000a, Part II - Table 14).

Standard Error ($ó_{â}$) The standard error was calculated as the average of the standard errors implied by the reported lower and upper bounds of the percent increase (Samet et al., 2000a, Part II - Table 14)

<u>Hospital Admissions for Pneumonia (Samet et al., 2000a, 14 Cities)</u>

The C-R function to estimate the change in hospital admissions for pneumonia associated with daily changes in PM is:

$$Ä \text{ Pneumonia Admissions} = - y_0 \ (e^{-âÄPM} - 1) \cdot pop$$

where:

y_0 = daily hospital admission rate for pneumonia per person 65 and older = 5.16 E-5

â = PM10 coefficient = 0.00207

ÄPM = change in daily average PM concentration

pop = population age 65 and older

$ó_{â}$ = standard error of â = 0.00058

Incidence Rate. Pneumonia hospital admissions (ICD-9 codes: 480-487) are based on "Patient Discharge Data 1998-1999," California Office of Statewide Health Planning and Development, 2000. Population data are from "Race/Ethnic Population with Age and Sex Detail, 1970-2040", California Department of Finance.

Coefficient Estimate (â). The coefficient is estimated from relative risk of 1.021 which is based on a 2.07 percent increase in admissions due to a PM10 change of 10.0 µg/m^3 (Samet et al., 2000a, Part II - Table 14).

Standard Error ($ó_{â}$). The standard error was calculated as the average of the standard errors implied by the reported lower and upper bounds of the percent increase (Samet et al., 2000a, Part II - Table 14)

Hospital Admissions for Cardiovascular Disease (Samet et al., 2000a, 14 Cities)

The C-R function to estimate the change in hospital admissions for cardiovascular disease associated with daily changes in PM10 is:

$$\Delta \text{CVD Admissions} = - y_0 \ (e^{-\hat{a}\Delta PM} - 1) \cdot pop$$

where:

y_0 = daily hospital admission rate for cardiovascular disease per person 65 and older = 1.58E-4

$\hat{a}$ = PM10 coefficient = 0.00119

ÄPM = change in daily average PM concentration

pop = population age 65 and older

$ó_{\hat{a}}$ = standard error of $\hat{a}$ = 0.00011

Incidence Rate. Congestive heart failure hospital admissions (ICD-9 codes: 390-429) are based on "Patient Discharge Data 1998-1999," California Office of Statewide Health Planning and Development, 2000. Population data are from "Race/Ethnic Population with Age and Sex Detail, 1970-2040", California Department of Finance.

Coefficient Estimate ($\hat{a}$). The coefficient is estimated from a relative risk of 1.012 which is based on a 1.19 percent increase in admissions due to a PM10 change of 10.0 µg/m^3 (Samet et al., 2000a, Part II - Table 14).

Standard Error ($ó_{\hat{a}}$). The standard error was calculated as the average of the standard errors implied by the reported lower and upper bounds of the percent increase (Samet et al., 2000a, Part II - Table 14)

Hospital Admissions for Asthma (Sheppard et al., 1999, Seattle)

Sheppard et al. (1999) studied the relation between air pollution in Seattle and non-elderly hospital admissions for asthma from 1987 to 1994. They used air quality data for PM10, PM2.5, coarse PM2.5-10, SO2, ozone, and CO in a Poisson regression model with controls for time trends, seasonal variations, and temperature-related weather effects. They found asthma hospital admissions associated with PM10, PM2.5, coarse PM2.5-10, CO, and ozone. The C-R function is based on a two-pollutant model with CO and PM2.5 and PM10 single-pollutant model:

$$\Delta \text{Asthma Admissions} = - y_0 \ (e^{-\hat{a}\Delta PM} - 1) \cdot pop$$

where:

y_0 = daily hospital admission rate for asthma per person = 2.63 E-6

$\hat{a}$ = PM2.5 coefficient = 0.002505, PM10 coefficient = 0.002568

ÄPM = change in daily average PM concentration

pop = population of ages less than 65

$ó_{\hat{a}}$ = standard error of PM2.5 $\hat{a}$ = 0.001045, standard error of PM10 $\hat{a}$ = 0.0007674

Incidence Rate. Hospital admissions for asthma (ICD-9 code: 493) are based on "Patient Discharge Data 1998-1999," California Office of Statewide Health Planning and

Development, 2000. Population data are from "Race/Ethnic Population with Age and Sex Detail, 1970-2040", California Department of Finance.

Coefficient Estimate (â). Based on a model with CO, the daily average coefficient is estimated from the relative risk (1.03) associated with a change in PM2.5 exposure of 11.8µg/m^3 (Sheppard et al., 1999, Table 3 and p. 28).

Standard Error ($ó_{â}$). The standard error was calculated as the average of the standard errors implied by the reported lower and upper bounds of the relative risk (Sheppard et al., 1999, p. 28).

Hospital Admissions for Circulatory, Chronic Respiratory, and Acute Respiratory Diseases (Van Den Eeden et al., 1999)

Three recent studies (Moolgavkar, 2000; Van Den Eeden et al., 1999; and Linn et al., 2000) in California have examined the relationship between changes in ambient PM concentrations and the number of daily circulatory or respiratory hospitalizations. All three focus on Southern California, but they use different subcategories of diseases and different age groupings. In general, they found statistically significant relationships between daily PM concentrations (measured as PM10 or as PM2.5) and daily hospital admissions for cardiovascular and respiratory illnesses. The Van Den Eeden study is the most comprehensive and covers all the endpoints included in all other studies, although they are grouped (ICD-9 codes) a little differently. It also covers all age population while other studies were only for certain age groups. We select this study as an alternative estimate for all hospital admissions. The C-R function is:

$$Ä \text{ Circulatory or Respiratory Admissions} = - y_0 \ (e^{-âÄPM} - 1) \cdot pop$$

where:

ÄPM= change in daily average PM concentration

pop= population of all ages, and other variables are shown as follows:

	Circulatory	Chronic Respiratory	Acute Respiratory
y_0 = daily hospital admission rate per person	3.33E-5	6.74E-6	1.60E-5
â = PM10 coefficient	0.000797	0.002700	0.001241
$ó_{â}$ = standard error of â	0.000389	0.000774	0.000468

Incidence Rate. Hospital admissions for circulatory (ICD-9 codes: 410-414, 428, 430-438, 415-417, 420-427, 429, 440-459), chronic respiratory (e.g., COPD, asthma, ICD-9 codes: 493, 490-492, 494-496), and acute respiratory (e.g., pneumonia, ICD-9 codes: 460-466,472-478, 480-487, 500-519) are based on "Patient Discharge Data 1998-1999,"

California Office of Statewide Health Planning and Development, 2000, and population data are from "Race/Ethnic Population with Age and Sex Detail, 1970-2040", California Department of Finance.

Coefficient Estimate (â). The coefficient is estimated from a relative risk of 1.008, 1.027, and 1.012 for a PM10 change of 10 $\mu g/m^3$.

Emergency Room Visits for Asthma (Schwartz et al., 1993, Seattle)

Schwartz et al. (1993) examined the relationship between air quality and emergency room visits for asthma in persons under 65, and 65 and over who lived in Seattle from September 1989 to September 1990. Using single-pollutant models they found daily levels of PM10 linked to ER visits in individuals younger than 65.

The C-R function to estimate the change in daily emergency room visits for asthma associated with daily changes in PM10 is:

$$\Ä \text{ Asthma ER Visits} = - y_0 \ (e^{-\hat{a}\Ä PM} - 1) \cdot pop$$

where:

y_0 = daily ER visits for asthma per person under 65 years old = 4.48 E-6

â = PM10 coefficient (Schwartz et al., 1993, p. 829) = 0.00367

ÄPM = change in daily average PM concentration

pop = population of ages 0-64

$ó_{\hat{a}}$ = standard error of â (Schwartz et al., 1993, p. 829) = 0.00126

Incidence Rate. Smith et al. (1997, p. 789) reported that in 1987 there were 445,000 asthma admissions and 1.2 million asthma ER visits. Assuming that all asthma hospital admissions pass through the ER room, then 37% of ER visits end up as hospital admissions. By subtracting out those visits that end up as admissions, ER visits = 1.7*asthma admission rate = 1.7*2.63 E-6 =4.48 E-6. Asthma hospital admissions (ICD-9 code: 493) rate are based on "Patient Discharge Data 1998-1999," California Office of Statewide Health Planning and Development, 2000, and population data are from "Race/Ethnic Population with Age and Sex Detail, 1970-2040", California Department of Finance.

10.1.5.7 Minor Illness

In addition to chronic illnesses and hospital admissions, there is considerable scientific research that has reported significant relationships between elevated air pollution levels and other morbidity effects. Controlled human studies have established relationships between air pollution and symptoms such as cough, pain on deep inspiration, wheeze, eye irritation and headache. In addition, epidemiological research has found relationships between air pollution exposure and acute infectious diseases (e.g., bronchitis, sinusitis) and a variety of "symptom-day" categories. Some "symptom-day" studies examine excess incidences of days with identified symptoms such as wheeze, cough, or other specific upper or lower respiratory symptoms. Other studies estimate relationships for days with a more general description of days with adverse health impacts, such as "respiratory restricted activity days" or work loss days.

We selected a few endpoints that reflect some minor morbidity effects and carefully adjusted estimates to avoid double counting (e.g., adjusted minor restricted activity days by number of asthma attacks).

Acute Bronchitis C-R Function (Dockery et al., 1996)

Dockery et al. (1996) examined the relationship between PM and other pollutants on the reported rates of asthma, persistent wheeze, chronic cough, and bronchitis, in a study of 13,369 children ages 8-12 living in 24 communities in the U.S. and Canada. Health data were collected in 1988-1991, and single-pollutant models were used in the analysis to test a number of measures of particulate air pollution. The study found that there was a marginally significant relationship between PM and bronchitis.

The C-R function to estimate the change in acute bronchitis is:

$$\Delta Acute\ Bronchitis = -[\frac{y_0}{(1-y_0)\cdot e^{\Delta PMb} + y_0} - y_0]\cdot pop$$

where:

Y_0= annual bronchitis incidence rate per person = 0.044

Â = estimated PM2.5 logistic regression coefficient = 0.0272

ÄPM = change in annual average PM concentration

pop = population of ages 8-12

$ó_{â}$ = standard error of â = 0.0171

Incidence Rate. The estimation of incidence rate is detailed in "Final Heavy Duty Engine/Diesel Fuel Rule: Air Quality Estimation, Selected Health and Welfare Benefits Methods, and Benefit Analysis Results, Appendix C", U.S. EPA, December 2000.

Coefficient Estimate (â). The estimated logistic coefficient is based on the odds ratio (= 1.50) associated with being in the most polluted city ($PM_{2.1}$= 20.7 µg/m^3) versus the least polluted city ($PM_{2.1}$=5.8 µg/ m^3) (Dockery et al., 1996, Tables 1 and 4). We applied the PM2.1 coefficient to PM2.5 and PM10.

Standard Error ($ó_{â}$) The standard error of the coefficient is calculated from the reported lower and upper bounds of the odds ratio (Dockery et al., 1996, Table 4)

Upper Respiratory Symptoms (Pope et al., 1991)

Using logistic regression, Pope et al. (1991) estimated the impact of PM10 on the incidence of a variety of minor symptoms in 55 subjects (34 "school-based" and 21 "patient-based") living in the Utah Valley from December 1989 through March 1990. The children in the Pope et al. study were asked to record respiratory symptoms in a daily diary. Pope et al. defined upper respiratory symptoms as consisting of one or more of the following symptoms: runny or stuffy nose; wet cough; and burning, aching, or red eyes. The sample in this study was relatively small and is most representative of the asthmatic population, rather than the general population. The school- based subjects (ages 9 to 11) were chosen based on "a positive response to one or more of three questions: ever wheezed without a cold, wheezed for 3 days or more out of the week for a month or longer, and/or had a doctor say the 'child has asthma' (Pope et al., 1991, p. 669)." The patient-based subjects (ages 8 to 72) were receiving treatment for asthma and were referred by local physicians. Regression results for the school-based sample (Pope et al., 1991, Table 5) showed PM10 significantly associated with both upper and lower respiratory symptoms. The patient-based sample did not find a significant PM10 effect. The results from the school-based sample are used here.

The C-R function used to estimate the change in upper respiratory symptoms is:

$$\Delta Upper\ Respiratory\ Symptoms = -[\frac{y_0}{(1-y_0)\cdot e^{\Delta PMb} + y_0} - y_0]\cdot pop$$

where:

y_0 = daily upper respiratory symptom incidence rate per person = 0.3419

â = estimated PM10 logistic regression coefficient = 0.0036 (Pope et al., 1991, Table 5)

ÄPM = change in daily average PM concentration

pop = asthmatic population ages 9 to 11 = 6.91% of population ages 9 to 11

$ó_{â}$ = standard error of â (Pope et al., 1991, Table 5) = 0.0015

Incidence Rate. The incidence rate is published in Pope et al. (Pope et al., 1991, Table 2). Taking a sample-size-weighted average, one gets an incidence rate of 0.3419.

Lower Respiratory Symptoms (Schwartz et al., 1994)

Schwartz et al. (1994) used logistic regression to link lower respiratory symptoms in children with SO_2, NO_2, ozone, PM10, PM2.5, sulfate and H^+ (hydrogen ion). Children were selected for the study if they were exposed to indoor sources of air pollution: gas stoves and parental smoking. The study enrolled 1,844 children in 1984 into a year-long study. The study was conducted in different years (1984 to 1988) in six cities. The students were in grades two through five at the time of enrollment in 1984. By the completion of the final study, the cohort would then be in the eighth grade (ages 13-14); this suggests an age range of 7 to 14.

In single pollutant models SO_2, NO_2, PM2.5, and PM10 were significantly linked to coughing. In two-pollutant models, PM10 had the most consistent relationship with coughing. In models for upper respiratory symptoms, they reported a marginally significant association for PM10. In models for lower respiratory symptoms, they reported significant single-pollutant models, using SO_2, O_3, PM2.5, PM10, SO4, and H^+.

The C-R function used to estimate the change in lower respiratory symptoms is:

$$\Delta Lower\ Respiratory\ Symptoms = -[\frac{y_0}{(1-y_0)\cdot e^{\Delta PMb} + y_0} - y_0]\cdot pop$$

where:

y_0 = daily lower respiratory symptom incidence rate per person = 0.0012

â = estimated PM2.5 logistic regression coefficient = 0.01823

ÄPM = change in daily average PM concentration

pop = population of ages 7-14

$ó_{â}$ = standard error of â = 0.00586

Incidence Rate. The proposed incidence rate, 0.12 percent, is based on the percentiles in Schwartz et al. (Schwartz et al., 1994, Table 2). The calculation is detailed in “Final Heavy Duty Engine/Diesel Fuel Rule: Air Quality Estimation, Selected Health and

Welfare Benefits Methods, and Benefit Analysis Results, Appendix C", U.S. EPA, December 2000.

Coefficient Estimate (â). The coefficient is calculated from the reported odds ratio (= 1.44) in a single-pollutant model associated with a 20 µg/m^3 change in PM2.5 (Schwartz et al., 1994, Table 5).

Standard Error ($ó_{â}$). The standard error for the coefficient is calculated from the reported lower and upper bounds of the odds ratio (Schwartz et al., 1994, Table 5).

Asthma Attacks, (Whittemore and Korn, 1980)

Whittemore and Korn (1980) examined the relationship between air pollution and asthma attacks in a survey of 443 children and adults, living in six communities in southern California during three 34-week periods in 1972-1975. The analysis focused on TSP and ozone. In a two-pollutants model, daily levels of both TSP and O_3 were significantly related to reported asthma attacks.

The C-R function to estimate the change in the number of asthma attacks is:

$$\Delta\ Asthma\ Attacks = -[\frac{y_0}{(1-y_0)\cdot e^{\Delta PMb} + y_0} - y_0]\cdot pop$$

where:

y_0 = daily incidence of asthma attacks = 0.027 (Krupnick, 1988, p. 4-6)

â = PM10 coefficient = 0.00144

ÄPM = change in daily PM concentration

pop = population of asthmatics of all ages = 5.61% of the population of all ages (Adams and Marano, 1995 Table 57).

$ó_{â}$ = standard error of â = 0.000556

Incidence Rate. The annual rate of 9.9 asthma attacks per asthmatic is divided by 365 to get a daily rate. A figure of 9.9 is roughly consistent with the recent statement that "People with asthma have more than [a combined] 100 million days of restricted activity" each year (National Heart, lung, and Blood Institute 1997, p. 1). This 100 million incidence figure coupled with the 1996 population of 265,557,000 (U.S. Bureau of the Census, 1997, Table 2) and the latest asthmatic prevalence rate of 5.61% (Adams and Marano, 1995, Table 57), suggest an annual asthma attack rate per asthmatic of 6.7.

Coefficient Estimate (â). Based on a model with ozone, the coefficient is based on a TSP coefficient (0.00079) (Whittemore and Korn, 1980, Table 5). Assuming that PM10 is 55 percent of TSP and that particulates greater than ten micrometers are harmless.

Standard Error ($ó_{â}$). The standard error is calculated from the two-tailed p-value (<0.01) reported by Whittemore and Korn (1980, Table 5), which implies a t-value of at least 2.576 (assuming a large number of degrees of freedom).

Work Loss Days (Ostro, 1987)

Ostro (1987) estimated the impact of PM2.5 on the incidence of work-loss days (WLDs), restricted activity days (RADs), and respiratory-related RADs (RRADs) in a national sample of the adult working population, ages 18 to 65, living in metropolitan areas. The annual national survey results used in this analysis were conducted in 1976-1981. Ostro

reported that two-week average PM2.5 levels were significantly linked to work-loss days, RADs, and RRADs, however there was some year-to-year variability in the results. Separate coefficients were developed for each year in the analysis (1976-1981); these coefficients were pooled. The coefficient used in the concentration-response function used here is a weighted average of the coefficients in Ostro (1987, Table III) using the inverse of the variance as the weight.

The C-R function to estimate the change in the number of work-loss days is:

$$\Ä \text{ WLD} = - y_0 \ (e^{-â\ÄPM} - 1) \cdot \text{pop}$$

where:

y_0 = daily work-loss-day incidence rate per person = 0.00648

â = inverse-variance weighted PM2.5 coefficient = 0.0046

ÄPM = change in daily average PM concentration

pop = population of ages 18 to 65

$ó_â$ = standard error of â = 0.00036

Incidence Rate. The estimated 1994 annual incidence rate is the annual number (376,844,000) of WLD per person in the age 18-64 population divided by the number of people in 18-64 population (159,361,000). The 1994 daily incidence rate is calculated as the annual rate divided by 365. Data are from U.S. Bureau of the Census (1997, Table 14) and Adams (1995, Table 41).

Coefficient Estimate (â). The coefficient used in the C-R function is a weighted average of the coefficients in Ostro (1987, Table III) using the inverse of the variance as the weight.

Standard Error ($ó_â$). The standard error of the coefficient calculation is detailed in “Final Heavy Duty Engine/Diesel Fuel Rule: Air Quality Estimation, Selected Health and Welfare Benefits Methods, and Benefit Analysis Results, Appendix C”, U.S. EPA, December 2000.

<u>Minor Restricted Activity Days (Ostro and Rothschild, 1989)</u>

Ostro and Rothschild (1989) estimated the impact of PM2.5 on the incidence of minor restricted activity days (MRADs) and respiratory-related restricted activity days (RRADs) in a national sample of the adult working population, ages 18 to 65, living in metropolitan areas. The annual national survey results used in this analysis were conducted in 1976-1981. Controlling for PM2.5, two-week average O_3 has highly variable association with RRADs and MRADs. Controlling for O_3, two-week average PM2.5 was significantly linked to both health endpoints in most years.

The study is based on a sample of individuals ages 18-65. Applying the C-R function to this age group is likely a slight underestimate, as it seems likely that the elderly are at least as susceptible to PM as individuals 65 and younger. The elderly appear more likely to die due to PM exposure than other age groups (e.g., Schwartz, 1994d, p. 30) and a number of studies have found that hospital admissions for the elderly are related to PM exposures (e.g., Schwartz, 1994b; Schwartz, 1994c).

The coefficient used in this analysis is a weighted average of the coefficients in Ostro and Rothschild (1989), Table 4, using the inverse of the variance as the weight. The C-R

function to estimate the change in the number of minor restricted activity days (MRAD) is:

$$Ä\ MRAD = - y_0\ (e^{-âÄPM} - 1) \cdot pop$$

where:

y_0 = daily MRAD daily incidence rate per person = 0.02137

â = inverse-variance weighted PM2.5 coeffcient = 0.00741

ÄPM = change in daily average PM concentration

pop = adult population ages 18 to 65

$ó_â$ = standard error of â = 0.0007

Incidence Rate. The annual incidence rate (7.8) provided by Ostro and Rothschild (1989, p. 243) was divided by 365 to get a daily rate of 0.02137.

Coefficient Estimate (â). The coefficient is a weighted average of the coefficients in Ostro and Rothschild (1989, Table 4) using the inverse of the variance as the weight.

10.1.6 Applicability of the C-R functions in California

Since many epidemiological studies do not incorporate results from California, one may expect that the health effects of particulate matter in California are different than those in the rest of the United States. One of reasons there may be differences is that the composition of particulate matter varies significantly by region, and it is possible that not all types of particulate matter have the same health effects. One obvious difference between particulate matter in California (and elsewhere in western states) and the rest of the country is that the sulfate aerosol content is much lower in California.

Samet et al. (2000a) provide data that allow a simple illustration of this difference. They report mean levels of several criteria air pollutants for 1987 to 1994 in 20 of the largest cities and metropolitan areas in the United States, including 6 in California: Los Angeles, San Diego, Santa Ana-Anaheim, San Jose, San Bernardino, and Oakland. Sulfur dioxide (SO_2) and nitrogen dioxide (NO_2) are gaseous pollutants, but they are precursors to the sulfate and nitrate aerosols that make up a significant share of PM10. Table 10.3 shows that PM10, ozone, and NO_2 are all somewhat higher, on average, in California cities than in other U.S. cities, with the largest difference in NO_2. SO_2, on the other hand, is dramatically lower in California cities. The slightly higher concentrations of PM10 and ozone in California cities reflects to some extent the warm temperatures and sunny skies that contribute to the photochemical formation of ozone and fine particulates. Dramatically lower SO_2 concentrations in California reflect that, to the extent that coal is burned by electric utilities and other industrial sources, it is low sulfur (western) coal that is used. Coal mined in the eastern United States, and widely used as a fuel for power plants and other industrial sources, tends to have substantially higher sulfur content, which has a direct relationship with ambient SO_2 concentrations.

Table 10.3. Comparison of mean concentrations of selected air pollutants, 1987-1994

	PM10 (µg/m³)	Ozone (ppb)	SO_2 (ppb)	NO_2 (ppb)
Six California cities	35.1	24.7	1.4	28.6
Fourteen other U.S. cities	31.9	22.3	6.7	22.3

Source: Samet et al., 2000a.

Although there has been substantial discussion in the literature of potential differences in health effects of various PM10 constituents, and some studies have reported that sulfate aerosols are more likely than other constituents to be a primary culprit, the findings regarding sulfate have not been consistent and there is sufficient evidence of PM10 health effects in locations (e.g., Los Angeles) where the sulfate content of PM10 is relatively low.

Numerous time-series studies provide opportunities to compare results obtained in California to those obtained in other locations in the United States. Comparing the results for PM10 in Table 10.2, the relative risks range from 1.003 to 1.020, with a mean value of 1.009. The weighted mean relative risk for all counties in California for PM10 is 1.008, with a 95% confidence interval of 1.004 to 1.012. This is within the range of mean results for studies throughout the United States, and suggests that the mortality effects of PM in California are comparable to those found in other locations in the United States.

Samet et al. (2000b) present the relative risk results for 20 cities in the United States, all estimated using the same estimation approach and years of data. They also estimate a pooled relative risk across all locations. The pooled relative risk for 10 µg/m³ PM10 results across all 20 locations was 1.005. Removing the California locations from these results and averaging the relative risk results from the remaining 14 city/counties results in an average relative risk value of 1.004. By comparison, the average relative risk for the six California locations was 1.009, and ranged from 1.003 to 1.020 across these six locations. This comparison suggests that the daily time-series results for PM10 from California are similar to, if not slightly higher than those from other locations across the country. These results contradict the hypothesis that PM health effects in California may be lower because of the significantly lower sulfate content of PM in the West.

Based on our observations, in cases where the EPA adopted C-R functions that do not incorporate results from California, or where the contribution from the California-based segment of the study population is unclear, the weight of available evidence from the other health outcome categories is not sufficient enough to argue that differences in the composition of the ambient PM in California or aspects of the California population make using results from locations outside of California inappropriate. We therefore selected functions which were drawn from the results of non-California locations when the California-specific C-R functions are not available.

10.2 Health Effects Results

Table 10.4 and Table 10.5 summarize the estimated health effects of reducing PM2.5 concentration from current levels to 12 ìg/m³ and to the non-anthropogenic background of 5 ìg/m³ in California.

Table 10.6 and Table 10.7 summarize the estimated health effects of reducing PM10 concentration from current levels to 20 ìg/m^3 and to the non-anthropogenic background of 10 ìg/m^3 in California.

Table 10.8 to Table 10.10 present the estimated mortality, chronic illness, and hospital admission effects of reducing PM2.5 concentration from current levels to the non-anthropogenic background in all California counties.

Table 10.4. California Annual PM2.5 Health Effects (Current Level Minus 12 μg/m³)

Health Endpoint	Reference	Estimated Beta (Standard Error)	Avoided Incidence (cases/year)		
			5th Percentile	Mean	95th Percentile
Mortality					
Long-Term Exposures Mortality					
Ages 30+	Krewski et al., 2000	0.0046257 (0.0012046)	3,229	6,526	9,754
Short-Term Exposures Mortality					
All Ages	Schwartz, 1996	0.001433 (0.000129)	1,604	1,945	2,286
Chronic Illness					
Chronic Bronchitis (Age 27+)	Abbey, 1995	0.0132 (0.00680)	-59	5,749	10,907
Hospitalization					
COPD (ICD codes 490-492, 494-496), Age 65+	Samet et al., 2000	0.002880 (0.001390)	33	600	1,154
Pneumonia (ICD codes 480-487), Age 65+	Samet et al., 2000	0.002070 (0.000580)	391	864	1,331
Cardiovascular (ICD codes 390-429), Age 65+	Samet et al., 2000	0.001190 (0.000110)	1,254	1,530	1,806
Asthma (ICD codes 493), Age 64-	Sheppard et al., 1999	0.002505 (0.001045)	86	470	846
Asthma-related ER Visits, Age 64-	Schwartz et al., 1993	0.003670 (0.001260)	386	1,167	1,930
Alternative Estimate					
Circulatory (ICD codes 410-414, 428, 430-438, 415-417, 420-427, 429, 440-459), All ages	Van Den Eeden et al., 1999, California	0.000797 (0.000389)	93	2,132	4,156
Chronic respiratory (e.g., COPD, asthma, ICD codes 493, 490-492, 494-496), All ages	Van Den Eeden et al., 1999, California	0.002700 (0.000774)	647	1,467	2,276
Acute respiratory (e.g., pneumonia ICD codes 460-466,472-478, 480-487, 500-519), All ages	Van Den Eeden et al., 1999, California	0.001241 (0.000468)	178	679	1,176
Minor Illness					
Acute Bronchitis, Age 8-12	Dockery et al., 1996	0.02720 (0.01710)	-4,663	17,473	34,149
URS, Age 9-11	Pope et al., 1991	0.00360 (0.0015)	38,371	208,384	376,874
LRS, Age 7-14	Schwartz et al., 1994	0.01823 (0.00586)	81,284	208,638	323,322
Asthma Attacks, All ages	Whittemore and Korn, 1980	0.00144 (0.000556)	41,390	169,381	296,178
Work Loss Days	Ostro, 1987	0.0046 (0.00036)	1,227,554	1,445,391	1,661,848
Minor Restricted Activity Days –adjusted*	Ostro & Rothschild, 1989	0.00741 (0.0007)	6,175,290	7,413,386	8,635,934

* To avoid double counting, the number of asthma attacks estimated were subtracted from the number of MRADs.

1

Table 10.5. California Annual PM2.5 Health Effects

(Current Level minus 9 ug/m3 for long-term mortality, and minus background Level for other endpoints)

Health Endpoint	Reference	Estimated Beta (Standard Error)	Avoided Incidence (cases/year)		
			5th Percentile	Mean	95th Percentile
Mortality					
Long-Term Exposures Mortality					
Ages 30+	Krewski et al., 2000	0.0046257 (0.0012046)	4,659	9,391	13,999
Short-Term Exposures Mortality					
All Ages	Schwartz, 1996	0.001433 (0.000129)	3,312	4,014	4,714
Chronic Illness					
Chronic Bronchitis (Age 27+)	Abbey, 1995	0.0132 (0.00680)	-122	11,414	20,918
Hospitalization					
COPD (ICD codes 490-492, 494-496), Age 65+	Samet et al., 2000	0.002880 (0.001390)	68	1,242	2,369
Pneumonia (ICD codes 480-487), Age 65+	Samet et al., 2000	0.002070 (0.000580)	814	1,791	2,751
Cardiovascular (ICD codes 390-429), Age 65+	Samet et al., 2000	0.001190 (0.000110)	2,608	3,180	3,750
Asthma (ICD codes 493), Age 64-	Sheppard et al., 1999	0.002505 (0.001045)	176	950	1,702
Asthma-related ER Visits, Age 64-	Schwartz et al., 1993	0.003670 (0.001260)	783	2,352	3,864
Alternative Estimate					
Circulatory (ICD codes 410-414, 428, 430-438, 415-417, 420-427, 429, 440-459), All ages	Van Den Eeden et al., 1999, California	0.000797 (0.000389)	189	4,343	8,450
Chronic respiratory (e.g., COPD, asthma, ICD codes 493, 490-492, 494-496), All ages	Van Den Eeden et al., 1999, California	0.002700 (0.000774)	1,317	2,973	4,592
Acute respiratory (e.g., pneumonia ICD codes 460-466,472-478, 480-487, 500-519), All ages	Van Den Eeden et al., 1999, California	0.001241 (0.000468)	363	1,381	2,386
Minor Illness					
Acute Bronchitis, Age 8-12	Dockery et al., 1996	0.02720 (0.01710)	-9,567	32,923	59,724
URS, Age 9-11	Pope et al., 1991	0.00360 (0.0015)	77,367	418,985	755,504
LRS, Age 7-14	Schwartz et al., 1994	0.01823 (0.00586)	160,279	398,777	600,088
Asthma Attacks, All ages	Whittemore and Korn, 1980	0.00144 (0.000556)	84,439	344,532	600,679
Work Loss Days	Ostro, 1987	0.0046 (0.00036)	2,487,857	2,923,535	3,354,714
Minor Restricted Activity Days –adjusted*	Ostro & Rothschild, 1989	0.00741 (0.0007)	12,439,319	14,873,148	17,257,232

To avoid double counting, the number of asthma attacks estimated were subtracted from the number of MRADs.

1

Table 10.6. California Annual PM10 Health Effects (Current Level minus 20 ug/m3)

Health Endpoint	Reference	Estimated Beta (Standard Error)	Avoided Incidence (cases/year)		
			5th Percentile	Mean	95th Percentile
Mortality					
Long-Term Exposures Mortality					
Ages 30+	Krewski et al., 2000	0.00231285 (0.0006023)*	3,236	6,533	9,753
Short-Term Exposures Mortality					
All Ages	Pooled California Studies (Chestnut & Mills, 2001)	0.000838 (0.000203)	1,210	2,295	3,373
Chronic Illness					
Chronic Bronchitis (Age 27+)	Abbey, 1993	0.00932 (0.00475)	10	7,850	14,500
Hospitalization					
COPD (ICD codes 490-492, 494-496), Age 65+	Samet et al., 2000	0.002880 (0.001390)	66	1,191	2,256
Pneumonia (ICD codes 480-487), Age 65+	Samet et al., 2000	0.002070 (0.000580)	785	1,721	2,636
Cardiovascular (ICD codes 390-429), Age 65+	Samet et al., 2000	0.001190 (0.000110)	2,514	3,063	3,611
Asthma (ICD codes 493), Age 64-	Sheppard et al., 1999	0.002568 (0.000767)	402	955	1,493
Asthma-related ER Visits, Age 64-	Schwartz et al., 1993	0.003670 (0.001260)	771	2,301	3,757
Alternative Estimate					
Circulatory (ICD codes 410-414, 428, 430-438, 415-417, 420-427, 429, 440-459), All ages	Van Den Eeden et al., 1999, California	0.000797 (0.000389)	187	4,270	8,292
Chronic respiratory (e.g., COPD, asthma, ICD codes 493, 490-492, 494-496), All ages	Van Den Eeden et al., 1999, California	0.002700 (0.000774)	1,294	2,909	4,476
Acute respiratory (e.g., pneumonia ICD codes 460-466,472-478, 480-487, 500-519), All ages	Van Den Eeden et al., 1999, California	0.001241 (0.000468)	357	1,356	2,338
Minor Illness					
Acute Bronchitis, Age 8-12	Dockery et al., 1996	0.02720 (0.01710)	-9,883	31,557	54,379
URS, Age 9-11	Pope et al., 1991	0.00360 (0.0015)	78,599	424,492	763,139
LRS, Age 7-14	Schwartz et al., 1994	0.01823 (0.00586)	160,586	389,225	572,660
Asthma Attacks, All ages	Whittemore and Korn, 1980	0.00144 (0.000556)	83,128	338,270	588,195
Work Loss Days	Ostro, 1987	0.0046 (0.00036)	2,399,490	2,814,815	3,224,423
Minor Restricted Activity Days -adjusted**	Ostro & Rothschild, 1989	0.00741 (0.0007)	11,933,013	14,215,093	16,435,564

* PM2.5 coeffecient and standard error were muliplied by 0.5 assuming only the PM 2.5 fraction of PM10 was associated with long-term mortality.

** To avoid double counting, the number of asthma attacks estimated were subtracted from the number of MRADs.

1

Table 10.7. California Annual PM10 Health Effects

(Current Level minus 18 ug/m3 for long-term mortality, and minus background level for other endpoints)

Health Endpoint	Reference	Estimated Beta (Standard Error)	Avoided Incidence (cases/year)		
			5th Percentile	Mean	95th Percentile
Mortality					
Long-Term Exposures Mortality					
Ages 30+	Krewski et al., 2000	0.00231285 (0.0006023)*	3,734	7,534	11,241
Short-Term Exposures Mortality					
All Ages	Pooled California Studies (Chestnut & Mills, 2001)	0.000838 (0.000203)	2,148	4,069	5,969
Chronic Illness					
Chronic Bronchitis (Age 27+)	Abbey, 1993	0.0032 (0.00475)	16	13,530	24,141
Hospitalization					
COPD (ICD codes 490-492, 494-496), Age 65+	Samet et al., 2000	0.002880 (0.001390)	118	2,112	3,967
Pneumonia (ICD codes 480-487), Age 65+	Samet et al., 2000	0.002070 (0.000580)	1,401	3,061	4,671
Cardiovascular (ICD codes 390-429), Age 65+	Samet et al., 2000	0.001190 (0.000110)	4,487	5,464	6,436
Asthma (ICD codes 493), Age 64-	Sheppard et al., 1999	0.002568 (0.000767)	703	1,664	2,586
Asthma-related ER Visits, Age 64-	Schwartz et al., 1993	0.003670 (0.001260)	1,349	3,992	6,465
Alternative Estimate					
Circulatory (ICD codes 410-414, 428, 430-438, 415-417, 420-427, 429, 440-459), All ages	Van Den Eeden et al., 1999, California	0.000797 (0.000389)	328	7,497	14,520
Chronic respiratory (e.g., COPD, asthma, ICD codes 493, 490-492, 494-496), All ages	Van Den Eeden et al., 1999, California	0.002700 (0.000774)	2,268	5,074	7,767
Acute respiratory (e.g., pneumonia ICD codes 460-466,472-478, 480-487, 500-519), All ages	Van Den Eeden et al., 1999, California	0.001241 (0.000468)	628	2,378	4,085
Minor Illness					
Acute Bronchitis, Age 8-12	Dockery et al., 1996	0.02720 (0.01710)	-17,452	50,335	80,421
URS, Age 9-11	Pope et al., 1991	0.00360 (0.0015)	135,810	730,815	1,308,545
LRS, Age 7-14	Schwartz et al., 1994	0.01823 (0.00586)	270,413	631,880	899,973
Asthma Attacks, All ages	Whittemore and Korn, 1980	0.00144 (0.000556)	146,184	592,736	1,027,020
Work Loss Days	Ostro, 1987	0.0046 (0.00036)	4,195,917	4,910,652	5,612,157
Minor Restricted Activity Days –adjusted**	Ostro & Rothschild, 1989	0.00741 (0.0007)	20,717,957	24,564,726	28,272,025

* PM2.5 coeffecient and standard error were muliplied by 0.5 assuming only the PM 2.5 fraction of PM10 was associated with long-term mortality.

** To avoid double counting, the number of asthma attacks estimated were subtracted from the number of MRADs.

Table 10.8. County Annual PM2.5 Mortality Effects (Current Level Minus Background Levels)

County	Concentration Change (Current minus 9 ug/m^3)	Long-term Exposure Mortality Krewski, 2000, 63 cities, Age 30+				Short-term Exposure Mortality Schwartz, 1996, All ages			
		Population (age 30+)	5th Percentile	Mean	95th Percentile	Population (all ages)	5th Percentile	Mean	95th Percentil
ALAMEDA	6.8	830,217	156	317	474	1,443,741	120	145	170
ALPINE	0	745	--	--	--	1,208	0	0	0
AMADOR	7.6	23,696	6	12	18	35,100	5	6	7
BUTTE	3.3	115,129	16	32	47	203,171	17	21	25
CALAVERAS	7.6	27,521	7	14	21	40,554	6	7	8
COLUSA	3.3	9,750	1	2	3	18,804	1	1	2
CONTRA COSTA	6.8	560,627	97	197	296	948,816	81	99	116
DEL NORTE	0	16,430	--	--	--	27,507	1	1	1
EL DORADO, Lake Tahoe Basin	0	20,358	--	--	--	32,795	1	1	1
EL DORADO, Mountain Counties Basin	7.6	76,670	42	85	128	123,603	12	15	17
FRESNO	13.3	398,493	155	311	463	799,407	104	126	147
GLENN	3.3	14,402	2	4	5	26,453	2	2	3
HUMBOLDT	0	73,435	--	--	--	126,518	3	4	5
IMPERIAL	4.1	73,048	7	15	23	142,361	7	9	10
INYO	0	11,785	--	--	--	17,945	1	1	1
KERN, Mojave Basin	1	57,133	2	3	5	112,480	4	5	6
KERN, San Joaquinn Valley Basin	13.3	278,942	110	220	328	549,165	75	91	106
KINGS	13.3	65,080	20	40	59	129,461	13	16	19
LAKE	0	38,073	--	--	--	58,309	0	0	0
LASSEN	0	19,716	--	--	--	33,828	0	0	0
LOS ANGELES, Mojave	1	152,395	4	9	13	285,580	10	12	14
LOS ANGELES, South Coast Basin	13.2	4,927,449	1,763	3,546	5,274	9,233,758	1,086	1,316	1,545
MADERA	13.3	66,083	25	50	74	123,109	17	21	25
MARIN	6.8	167,482	27	55	82	247,289	22	26	31
MARIPOSA	7.6	11,432	3	6	9	17,130	2	3	3
MENDOCINO	0	52,390	--	--	--	86,265	2	3	3
MERCED	13.3	102,065	39	79	117	210,554	27	32	38
MODOC	0	6,043	--	--	--	9,449	0	0	0
MONO	0	7,604	--	--	--	12,853	0	0	0
MONTEREY	0	211,980	--	--	--	401,762	7	8	10
NAPA	6.8	75,990	18	37	56	124,279	15	18	21
NEVADA	7.6	61,115	15	30	44	92,033	11	14	16
ORANGE	13.2	1,576,527	475	956	1,422	2,846,289	324	393	461
PLACER, Lake Tahoe Basin	0	6,033	--	--	--	9,936	0	0	0
PLACER, Sac Valley Basin	3.3	144,794	12	25	38	238,463	15	18	22
PLUMAS	7.6	14,018	4	7	11	20,824	3	3	4
RIVERSIDE, Mojave Basin	1	16,644	1	1	2	30,908	1	2	2
RIVERSIDE, Salton Sea Basin	4.1	166,438	22	44	66	309,077	23	27	32
RIVERSIDE, South Coast Basin	13.2	649,109	267	538	800	1,205,400	186	226	265
SACRAMENTO	3.3	680,201	65	132	199	1,223,499	75	91	107
SAN BENITO	0	27,492	--	--	--	53,234	1	1	1
SAN BERNARDINO, Mojave Basin	1	197,817	5	11	17	393,170	14	17	21
SAN BERNARDINO, South Coast Basin	13.2	662,256	236	475	706	1,316,264	165	200	234
SAN DIEGO	6.6	1,550,162	276	560	839	2,813,833	222	269	316
SAN FRANCISCO	6.8	503,126	100	203	304	776,733	78	94	111
SAN JOAQUIN	13.3	294,878	125	251	374	563,598	83	101	119
SAN LUIS OBISPO	2.8	145,609	13	26	38	246,681	16	19	23
SAN MATEO	6.8	432,917	74	150	225	707,161	58	71	83

County	Concentration Change (Current minus 9 ug/m^3)	Long-term Exposure Mortality Krewski, 2000, 63 cities, Age 30+				Short-term Exposure Mortality Schwartz, 1996, All ages			
		Population (age 30+)	5th Percentile	Mean	95th Percentile	Population (all ages)	5th Percentile	Mean	95th Percentil
SANTA BARBARA	2.8	218,917	18	37	56	399,347	22	26	31
SANTA CLARA	6.8	960,713	134	271	406	1,682,585	107	130	153
SANTA CRUZ	0	146,100	--	--	--	255,602	5	6	7
SHASTA	3.3	98,835	12	25	38	163,256	13	16	19
SIERRA	7.6	2,400	1	1	2	3,555	1	1	1
SISKIYOU	0	28,852	--	--	--	44,301	0	0	0
SOLANO, Sac Valley Basin	3.3	67,412	5	11	16	121,998	6	7	9
SOLANO, San Francisco Basin	6.8	150,047	24	49	73	271,544	20	24	28
SONOMA, North Coast Basin	0	33,209	--	--	--	55,034	1	2	2
SONOMA, San Francisco Basin	6.8	243,531	49	99	149	403,580	41	49	58
STANISLAUS	13.3	233,429	96	193	287	446,997	65	79	93
SUTTER	3.3	43,620	5	9	14	78,930	5	6	8
TEHAMA	3.3	33,278	4	9	14	56,039	5	6	7
TRINITY	0	8,872	--	--	--	13,022	0	0	1
TULARE	13.3	179,625	72	145	216	368,021	48	58	68
TUOLUMNE	7.6	36,235	9	18	27	54,501	7	8	10
VENTURA	2.8	419,350	28	57	86	753,197	35	42	50
YOLO	3.3	83,401	7	15	23	168,660	9	11	13
YUBA	3.3	31,142	4	8	12	60,219	4	5	6
Statewide Total		18,640,255	4,659	9,391	13,999	33,870,743	***3,312***	***4,014***	***4,714***

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1 Table 10.9. County Annual PM2.5 Chronic Illness Effects (Current Level Minus Background Level)

County	**Concentration Change (Current minus 5 ug/m^3)**	**Chronic Bronchitis** **Abbey, 1995, Age 27+** **Est. â (std. Error) 0.0132 (0.00680)**			
		Population (age 27+)	5th Percentile	Mean	95th Percentile
ALAMEDA	10.80	902,538	-4	429	804
ALPINE	3.50	781	0	0	0
AMADOR	11.60	24,742	0	13	23
BUTTE	7.30	122,055	0	40	77
CALAVERAS	11.60	28,456	0	14	27
COLUSA	7.30	10,459	0	3	7
CONTRA COSTA	10.80	598,543	-3	285	533
DEL NORTE	2.50	17,621	0	2	4
EL DORADO, Lake Tahoe Basin	2.50	21,343	0	2	5
EL DORADO, Mountain Counties Basin	11.60	80,377	0	41	76
FRESNO	17.30	432,034	-3	316	569
GLENN	7.30	15,362	0	5	10
HUMBOLDT	2.50	78,240	0	9	18
IMPERIAL	8.10	79,320	0	29	55
INYO	3.50	12,232	0	2	4
KERN, Mojave Basin	5.00	61,888	0	14	28
KERN, San Joaquinn Valley Basin	17.30	302,161	-2	221	398
KINGS	17.30	72,019	-1	53	95
LAKE	-	39,676	0	0	0
LASSEN	-	21,551	0	0	0
LOS ANGELES, Mojave	5.00	166,631	0	38	74
LOS ANGELES, South Coast Basin	17.20	5,387,730	-42	3,915	7,062
MADERA	17.30	71,142	-1	52	94
MARIN	10.80	177,086	-1	84	158
MARIPOSA	11.60	11,917	0	6	11
MENDOCINO	2.50	55,283	0	6	13
MERCED	17.30	110,558	-1	81	146
MODOC	-	6,307	0	0	0

County	**Concentration Change (Current minus 5 ug/m^3)**	**Chronic Bronchitis** **Abbey, 1995, Age 27+** **Est. â (std. Error) 0.0132 (0.00680)**			
MONO	3.50	8,184	0	1	3
MONTEREY	2.50	231,186	0	27	53
NAPA	10.80	80,659	0	38	72
NEVADA	11.60	63,523	0	32	60
ORANGE	17.20	1,716,424	-14	1,247	2,250
PLACER, Lake Tahoe Basin	2.50	6,384	0	1	1
PLACER, Sac Valley Basin	7.30	153,220	-1	50	97
PLUMAS	11.60	14,517	0	7	14
RIVERSIDE, Mojave Basin	5.00	17,869	0	4	8
RIVERSIDE, Salton Sea Basin	8.10	178,691	-1	65	124
RIVERSIDE, South Coast Basin	17.20	696,897	-5	506	913
SACRAMENTO	7.30	734,152	-2	241	462
SAN BENITO	2.50	29,827	0	3	7
SAN BERNARDINO, Mojave Basin	5.00	214,586	0	49	95
SAN BERNARDINO, South Coast Basin	17.20	718,396	-6	522	942
SAN DIEGO	10.60	1,683,170	-8	786	1,476
SAN FRANCISCO	10.80	557,251	-3	265	497
SAN JOAQUIN	17.30	317,540	-3	232	418
SAN LUIS OBISPO	6.80	154,062	0	47	91
SAN MATEO	10.80	466,554	-2	222	416
SANTA BARBARA	6.80	235,598	-1	72	139
SANTA CLARA	10.80	1,050,455	-5	499	936
SANTA CRUZ	2.50	157,118	0	18	36
SHASTA	7.30	103,888	0	34	65
SIERRA	11.60	2,494	0	1	2
SISKIYOU	-	29,957	0	0	0
SOLANO, Sac Valley Basin	7.30	72,607	0	24	46
SOLANO, San Francisco Basin	10.80	161,609	-1	77	144
SONOMA, North Coast Basin	2.50	35,309	0	4	8
SONOMA, San Francisco Basin	10.80	258,935	-1	123	231
STANISLAUS	17.30	251,693	-2	184	331

County	**Concentration Change (Current minus 5 ug/m^3)**	**Chronic Bronchitis** **Abbey, 1995, Age 27+** **Est. â (std. Error) 0.0132 (0.00680)**			
SUTTER	7.30	46,746	0	15	29
TEHAMA	7.30	35,101	0	12	22
TRINITY	2.50	9,180	0	1	2
TULARE	17.30	194,596	-2	142	256
TUOLUMNE	11.60	37,927	0	19	36
VENTURA	6.80	450,600	-1	138	266
YOLO	7.30	90,504	0	30	57
YUBA	7.30	33,514	0	11	21
Statewide Total		20,208,974	-122	11,414	20,918

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1 Table 10.10. County Annual PM2.5 Hospitalization (Current Minus Background Levels)

County	Concentration Change (ug/m^3)	Population (age 65+)	COPD (ICD codes 490-492, 494-496) Samet et al., 2000, Age 65+ 0.002880 (0.001390)			Pneumonia (ICD codes 480-487) Samet et al., 2000, Age 65+ 0.002070 (0.000580)			Cardiovascular (ICD codes 390-429) Samet et al., 2000, Age 65+ 0.001190 (0.000110)			Asthma (ICD codes 493) Sheppard et al., 1999, Age 64- 0.002270 (0.000948)			
			5th %tile	Mean	95th %tile	5th %tile	Mean	95th %tile	5th %tile	Mean	95th %tile	Population (age 64-)	5th %tile	Mean	95th %tile
ALAMEDA	10.80	147,591	2	43	82	120	145	170	89	109	129	1,296,150	6	33	60
ALPINE	3.50	120	0	0	0	0	0	0	0	0	0	1,088	0	0	0
AMADOR	11.60	6,329	0	2	4	5	6	7	4	5	6	28,771	0	1	1
BUTTE	7.30	32,056	0	6	12	17	21	25	13	16	19	171,115	1	3	5
CALAVERAS	11.60	7,373	0	2	4	6	7	8	5	6	7	33,181	0	1	2
COLUSA	7.30	2,135	0	0	1	1	1	2	1	1	1	16,669	0	0	1
CONTRA COSTA	10.80	107,272	2	31	59	81	99	116	65	79	93	841,544	4	22	39
DEL NORTE	2.50	3,448	0	0	0	1	1	1	0	1	1	24,059	0	0	0
EL DORADO, Lake Tahoe Basin	2.50	4,057	0	0	1	1	1	1	1	1	1	28,738	0	0	0
EL DORADO, Mountain Counties Basin	11.60	15,277	0	5	9	11	13	15	10	12	14	108,227	1	3	5
FRESNO	17.30	79,209	2	36	69	137	166	195	77	93	110	720,198	5	29	52
GLENN	7.30	3,431	0	1	1	2	2	3	1	2	2	23,022	0	0	1
HUMBOLDT	2.50	15,776	0	1	2	2	2	3	2	3	3	110,742	0	1	1
IMPERIAL	8.10	14,305	0	3	6	14	17	20	7	8	9	128,056	0	2	4
INYO	3.50	3,429	0	0	1	0	1	1	1	1	1	14,516	0	0	0
KERN, Mojave Basin	5.00	10,549	0	1	3	3	4	5	3	4	4	101,930	0	1	2
KERN, San Joaquinn Valley Basin	17.30	51,505	1	24	45	114	138	162	50	61	72	497,661	4	20	36
KINGS	17.30	9,557	0	4	8	15	18	21	9	11	13	119,904	1	5	9
LAKE	--	11,359	--	--	--	0	0	0	0	0	0	46,950	0	0	0
LASSEN	--	3,054	--	--	--	0	0	0	0	0	0	30,774	0	0	0
LOS ANGELES, Mojave	5.00	27,800	0	4	7	12	14	17	8	10	11	257,780	1	3	6
LOS ANGELES, South Coast Basin	17.20	898,873	23	410	780	1,913	2,318	2,721	863	1,053	1,241	8,334,885	63	337	603
MADERA	17.30	13,596	0	6	12	21	25	29	13	16	19	109,513	1	4	8
MARIN	10.80	33,432	1	10	19	20	24	28	20	25	29	213,857	1	5	10
MARIPOSA	11.60	2,940	0	1	2	2	3	3	2	2	3	14,190	0	0	1
MENDOCINO	2.50	11,709	0	1	2	1	1	1	2	2	2	74,556	0	0	1

County	Concentration Change (ug/m^3)	Population (age 65+)	COPD (ICD codes 490-492, 494-496) Samet et al., 2000, Age 65+ 0.002880 (0.001390)			Pneumonia (ICD codes 480-487) Samet et al., 2000, Age 65+ 0.002070 (0.000580)			Cardiovascular (ICD codes 390-429) Samet et al., 2000, Age 65+ 0.001190 (0.000110)			Asthma (ICD codes 493) Sheppard et al., 1999, Age 64- 0.002270 (0.000948)			
			5th %tile	Mean	95th %tile	5th %tile	Mean	95th %tile	5th %tile	Mean	95th %tile	Population (age 64-)	5th %tile	Mean	95th %tile
MERCED	17.30	20,004	1	9	17	25	30	35	19	24	28	190,550	1	8	14
MODOC	--	1,663	--	--	--	0	0	0	0	0	0	7,786	0	0	0
MONO	3.50	976	0	0	0	0	1	1	0	0	0	11,877	0	0	0
MONTEREY	2.50	40,299	0	3	5	7	8	10	6	7	8	361,463	0	2	4
NAPA	10.80	19,086	0	6	11	12	14	17	12	14	17	105,193	0	3	5
NEVADA	11.60	16,049	0	5	10	13	15	18	10	13	15	75,984	0	2	4
ORANGE	17.20	280,763	7	128	244	440	533	625	270	329	388	2,565,526	19	104	186
PLACER, Lake Tahoe Basin	2.50	1,302	0	0	0	0	0	0	0	0	0	8,634	0	0	0
PLACER, Sac Valley Basin	7.30	31,258	0	6	12	11	13	15	13	16	18	207,205	1	4	7
PLUMAS	11.60	3,725	0	1	2	2	2	3	2	3	3	17,099	0	0	1
RIVERSIDE, Mojave Basin	5.00	3,919	0	1	1	1	1	2	1	1	2	26,988	0	0	1
RIVERSIDE, Salton Sea Basin	8.10	39,192	0	9	16	23	28	33	18	22	26	269,885	1	5	9
RIVERSIDE, South Coast Basin	17.20	152,850	4	70	133	178	215	252	147	179	211	1,052,550	8	43	76
SACRAMENTO	7.30	135,875	1	27	51	84	102	120	56	68	80	1,087,624	3	19	34
SAN BENITO	2.50	4,315	0	0	1	1	1	1	1	1	1	48,919	0	0	1
SAN BERNARDINO, Mojave Basin	5.00	33,686	0	5	9	16	19	22	9	12	14	359,484	1	4	8
SAN BERNARDINO, South Coast Basin	17.20	112,773	3	51	98	133	161	189	108	132	156	1,203,491	9	49	87
SAN DIEGO	10.60	313,750	5	89	171	215	260	306	186	227	268	2,500,083	12	63	113
SAN FRANCISCO	10.80	106,111	2	31	59	93	112	132	64	78	92	670,622	3	17	31
SAN JOAQUIN	17.30	59,799	2	27	52	137	166	194	58	70	83	503,799	4	21	37
SAN LUIS OBISPO	6.80	35,685	0	7	13	20	24	28	14	17	20	210,996	1	3	6
SAN MATEO	10.80	88,085	1	25	49	52	63	74	53	65	77	619,076	3	16	29
SANTA BARBARA	6.80	50,765	1	9	18	25	31	36	19	24	28	348,582	1	6	10
SANTA CLARA	10.80	160,527	3	46	89	154	187	219	97	118	140	1,522,058	7	39	70
SANTA CRUZ	2.50	25,487	0	2	3	6	7	8	4	4	5	230,115	0	1	3
SHASTA	7.30	24,861	0	5	9	15	18	21	10	12	15	138,395	0	2	4
SIERRA	11.60	629	0	0	0	0	1	1	0	0	1	2,926	0	0	0

County	Concentration Change (ug/m^3)	Population (age 65+)	COPD (ICD codes 490-492, 494-496) Samet et al., 2000, Age 65+ 0.002880 (0.001390)			Pneumonia (ICD codes 480-487) Samet et al., 2000, Age 65+ 0.002070 (0.000580)			Cardiovascular (ICD codes 390-429) Samet et al., 2000, Age 65+ 0.001190 (0.000110)			Asthma (ICD codes 493) Sheppard et al., 1999, Age 64- 0.002270 (0.000948)			
			5th %tile	Mean	95th %tile	5th %tile	Mean	95th %tile	5th %tile	Mean	95th %tile	Population (age 64-)	5th %tile	Mean	95th %tile
SISKIYOU	--	8,040	--	--	--	0	0	0	0	0	0	36,261	0	0	0
SOLANO, Sac Valley Basin	7.30	11,292	0	2	4	10	12	14	5	6	7	110,706	0	2	3
SOLANO, San Francisco Basin	10.80	25,134	0	7	14	20	24	28	15	19	22	246,410	1	6	11
SONOMA, North Coast Basin	2.50	6,957	0	0	1	1	1	1	1	1	1	48,076	0	0	1
SONOMA, San Francisco Basin	10.80	51,020	1	15	28	42	51	60	31	38	44	352,561	2	9	16
STANISLAUS	17.30	46,697	1	21	41	0	0	0	45	55	65	400,300	3	16	29
SUTTER	7.30	9,755	0	2	4	0	0	0	4	5	6	69,175	0	1	2
TEHAMA	7.30	8,923	0	2	3	0	0	0	4	4	5	47,116	0	1	1
TRINITY	2.50	2,241	0	0	0	0	0	0	0	0	0	10,781	0	0	0
TULARE	17.30	35,917	1	16	31	0	0	0	35	42	50	332,104	3	14	24
TUOLUMNE	11.60	10,067	0	3	6	0	0	0	7	8	9	44,434	0	1	2
VENTURA	6.80	76,804	1	14	27	0	0	0	29	36	42	676,393	2	11	20
YOLO	7.30	15,782	0	3	6	0	0	0	6	8	9	152,878	0	3	5
YUBA	7.30	6,410	0	1	2	0	0	0	3	3	4	53,809	0	1	2
Statewide Total		3,594,655	68	1,242	2,369	4,231	5,128	6,021	2,608	3,180	3,750	30,275,990	176	950	1,702

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